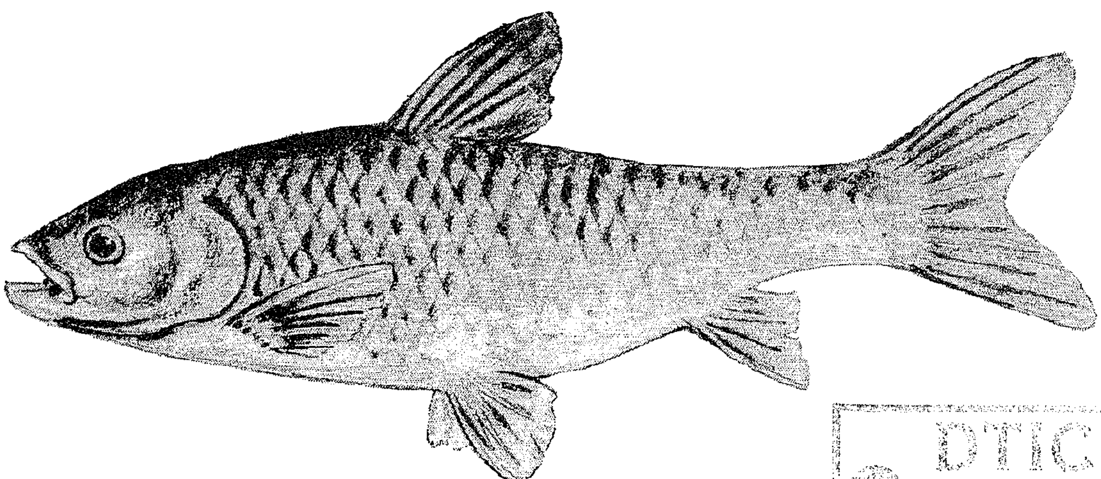
Proceedings of the Grass Carp Symposium

March 7-9, 1994
Gainesville, Florida



Proceedings of the Grass Carp Symposium

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Gainesville, Florida

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Acknowledgments

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Florida Game and Fresh Water Fish Commission

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IFAS Center for Aquatic Plants

The efforts and contributions by these agencies and those attending from other organizations ensured the success and quality of this workshop and are hereby recognized with gratitude.

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Agenda

Monday, March 7, 1994

7:30 a.m.	*Registration and Coffee*
8:30 a.m.	Announcements - W. T. Haller, Moderator
8:35 a.m.	Welcome. Dr. J. C. Joyce, Assoc. Vice-President for Agriculture and Natural Resources, Institute of Food and Agricultural Sciences.
8:40 a.m.	Why Are We Here? Mr. J. Ben Rowe, Commissioner, Florida Game and Fresh Water Fish Commission, Gainesville, FL.
8:50 a.m.	State Policies, Concerns and Issues Relative to Use of the Grass Carp. 3-5 min/speaker. California - Randall Stocker Florida - Dave Eggeman - Duke Hammond Louisiana - Louis Richardson North Carolina - Stratford Kay South Carolina - Steve de Kozlowski Texas - Phil Durocher Washington - Scott Bonar Virginia - Ed Steinkoenig
9:40 a.m.	*Break*

Introductory Session
Randall Stocker, Moderator

10:00 a.m.	History of the Grass Carp in Arkansas (United States). Don Fiegel, Arkansas Game and Fish Commission, Lonoke, AR. (Video)
10:15 a.m.	Grass Carp Movement in Texas Reservoirs. Earl Chilton II, Texas Parks and Wildlife, Austin, TX.
10:30 a.m.	Current and Future Objectives for Evaluation of Grass Carp Impacts on the Lower Trinity River and Galveston Bay. Mark Webb and Earl Chilton II, Texas Parks and Wildlife, Bryan and Austin, TX, respectively.
10:45 a.m.	Movement of Grass Carp in Large Florida Lakes, David Clapp, Illinois Natural History Survey, Charleston, IL.
11:00 a.m.	Grass Carp in the Pacific Northwest - Several Case Histories and Current Research. Scott Bonar, Washington Department of Wildlife, Olympia, WA.
11:25 a.m.	Status of Insect Biocontrol Against Hydrilla and Eurasian Watermilfoil. Gary Buckingham, USDA/ARS, Gainesville, FL, and Al Cofrancesco, U.S. Army Corps of Engineers, WES, Vicksburg, MS.
12:30 p.m.	*Lunch*

Recapture and Removal Session
Rue Hestand, Moderator

1:30 p.m. Removal of Grass Carp from Clear Lake. Lawson Snyder, Florida Game and Fresh Water Fish Commission, Kissimmee, FL.

1:45 p.m. Developments of Rotenone as an Oral Toxicant. Jim Fajt, Department of Fisheries, Auburn, AL.

2:00 p.m. Field Testing of Oral Rotenone Bait in Florida. Craig Mallison, Florida Game and Fresh Water Fish Commission, Eustis, FL.

2:15 p.m. Grass Carp Removal by Public Angling. Arnie Lingle, Florida Game and Fresh Water Fish Commission, Eustis, FL.

2:30 p.m. Potential Use of Acoustic Fish Barriers to Contain Grass Carp - A Research Need. Mike Curtin, Sonalysts, Inc., Waterford, CT. (Video)

2:45 p.m. Discussion

3:00 p.m. *Break*

Stocking Models and Potential Impacts
Greg Jubinsky, Moderator

3:30 p.m. Empirical Approach to Grass Carp Stocking Rates. Scott Bonar, Washington Department of Wildlife, Olympia, WA.

3:45 p.m. Utilization of WES Grass Carp Stocking Model. Mike Stewart and Will Boyd, U.S. Army Corps of Engineers, WES, Vicksburg, MS.

4:00 p.m. Aquatic Vegetation and Water Quality in Lake Marion, SC. Jeffrey Foltz, Department of Fisheries and Wildlife, Clemson University, Clemson, SC.

4:15 p.m. Discussion

Tuesday, March 8, 1994

Florida Case Histories
Jim Estes, Moderator

8:00 a.m. Fish Populations and Water Quality in Lakes Pearl and Baldwin. Doug Colle, Department of Fisheries and Aquatic Sciences, Gainesville, FL.

8:15 a.m. Effects of Grass Carp on the Aquatic Vegetation in Lake Conway. Andrew Leslie, Florida Department of Environmental Protection, Tallahassee, FL.

8:30 a.m. Largemouth Bass Response to Aquatic Plant Management in Central Florida Lakes. Wes Porak, Florida Game and Fresh Water Fish Commission, Eustis, FL.

8:45 a.m. The Effects of Triploid Grass Carp and Sonar Treatments on Aquatic Plants in Lake Yale. Rue Hestand, Florida Game and Fresh Water Fish Commission, Eustis, FL.

9:00 a.m. Fluctuations of Fish Populations in Lake Yale from 1987 to 1993. Craig Mallison, Florida Game and Fresh Water Fish Commission, Eustis, FL.

9:15 a.m. Utilization of Grass Carp for Milfoil Control in Deer Point Lake, FL. Jess Van Dyke, Florida Department of Environmental Protection, Tallahassee, FL.

9:30 a.m. Impacts of Grass Carp in Florida - A Case History of Johns Lake. Bruce Jaggers, Florida Game and Fresh Water Fish Commission, Eustis, FL.

9:45 a.m. Preliminary Results of the Lake Istokpoga Stocking or Triploid Carp - Effects on the Plants and Fishery. Dave Eggeman, Florida Game and Fresh Water Fish Commission, Tallahassee, FL.

10:00 a.m. *Break*

10:30 a.m. Discussion

11:30 a.m. *Lunch*

Texas and South Carolina
Alison Fox, Moderator

1:00 p.m. Effects of Macrophyte Removal by Crass Carp on Sportfish in Lake Conroe - The Early Years. Mike Maceina, Auburn University, Auburn, AL.

1:15 p.m. Status of Lake Conroe Fisheries - 12 Years of Macrophyte Removal. Mark Webb, Texas Parks and Wildlife, Bryan, TX.

1:30 p.m. Stocking Update and Vegetation Changes in Lake Marion, SC. Steve de Kozlowski, South Carolina Water Resources Commission, Columbia, SC.

1:45 p.m. Movements and Habitat Use of Triploid Grass Carp in Lake Marion, SC. Jeffrey Foltz, Department of Fisheries and Wildlife, Clemson University, Clemson, SC.

2:05 p.m. Considerations for Waterfowl in Submersed Aquatic Vegetation Control Programs. David Gordon, Ducks Unlimited, Georgetown, SC.

2:20 p.m. Discussion

2:45 p.m. *Break*

Lake Guntersville
A. Leon Bates, Moderator

3:00 p.m. Aquatic Vegetation in Guntersville Reservoir Following Grass Carp Stocking. David Webb, Tennessee Valley Authority, Muscle Shoals, AL.

3:20 p.m. Largemouth Bass Populations in Guntersville Reservoir Based on Creel Assessment. William Wrenn, Tennessee Valley Authority, Muscle Shoals, AL.

3:40 p.m. Collection, Age and Growth of Grass Carp in Lake Guntersville and the Santee Cooper System. Phil Kirk, J. V. Morrow, and K. J. Killgore, U.S. Army Corps of Engineers, WES, Vicksburg, MS.

4:00 p.m. Grass Carp Herbivory at Guntersville Reservoir, Alabama, and Its Impact on Waterfowl - Preliminary Results. Joseph Benedict, Auburn University, Auburn, AL.

4:15 p.m. Assessment of Grass Carp Reproduction in Guntersville Reservoir. Donny Lowery, Tennessee Valley Authority, Muscle Shoals, AL.

4:30 p.m.	Climate and Vegetation Impacts on Largemouth Bass in Lake Guntersville. Mike Maceina, Auburn University, Auburn, AL.
4:45 p.m.	A National Perspective. Lewis Decell, U.S. Army Corps of Engineers, WES, Vicksburg, MS.
5:15 p.m.	Bus leaves for Austin Cary Forest
7:00 p.m.	Barbecue

Wednesday, March 9, 1994

Orange and Lochloosa Lakes
Jim Estes, Moderator

8:30 a.m.	History of Hydrilla Control in the Orange and Lochloosa Lakes. Joe Hinkle, Florida Department of Environmental Protection, Lake City, FL.
9:00 a.m.	Organic Matter Deposition as a Result of Aquatic Weed Control Programs. Ken Langeland, Agronomy Department, Center for Aquatic Plants, Gainesville, FL.
9:15 a.m.	Littoral Muck Removal in Upper Kissimmee Chain of Lakes. Mike Hulon, Florida Game and Fresh Water Fish Commission, Kissimmee, FL.
9:30 a.m.	Grass Carp in Orange and Lochloosa - A Possible Scenario. Bill Haller, Agronomy Department, Center for Aquatic Plants, Gainesville, FL.
9:45 a.m.	Discussion

Closing Remarks

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Conversion Factors, Non-SI to SI Units of Measurement

Non-SI units of measurement used in this report can be converted to SI units as follows:

Multiply	By	To Obtain
acres	4,046.873	square meters
acre-feet	1,233.489	cubic meters
feet	0.3048	meters
gallons (U.S. liquid)	3.785412	liters
inches	2.54	centimeters
miles (U.S. statute)	1.609347	kilometers
pounds (mass)	0.4535924	kilograms
square miles	2.589998	square kilometers
yards	0.9144	meters

Introduction

by

J. L. Decell[1]

Throughout the United States, the use of the white amur (grass carp) has historically caused varying degrees of controversy, depending on the nature and location of the problem, the level of environmental sensitivity, policies, and politics.

There have been, and should remain, concerns over the use of this biological control method. Consistently, regardless of the motivating interest, one overriding concern has been the possible eradication of the submersed plants that provide habitat for fisheries. This is a continual concern, regardless of the method of control being employed. The use of the grass carp elevates this concern because of the perception that its use is commensurate with a lack of control over the degree and rate of control.

Compared with other methods, the use of the grass carp requires a significantly greater degree of understanding about both the agent and the environment in which it is placed to obtain the desired results. Nonetheless, the protection of the aquatic habitat should remain a part of the overall objective of any plant control operation.

A symposium was held in Gainesville on March 7-9, 1994, to discuss the effects of stocking grass carp for aquatic plant control. The purpose of the workshop was to provide information related to the concern for habitat protection and the suitability of using grass carp for aquatic plant control in large lakes. Individual lake studies, long-term impacts, low stocking rates, and grass carp recapture methods were presented in a context of using grass carp in a manner that prevents eradication of submersed aquatic plants. In addition, public perception and questions for experts were presented during panel discussions. Invited speakers were given a tour of area lakes.

As with all technology, it is necessary to assess the status of not only the state of the art, but the general view of its applications in the real-world environments. It was intended that this workshop serve at least a portion of that assessment. It is anticipated that another will serve the same need again in the future.

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1 Program Manager, Environmental Resources Research and Assistance Programs, U.S. Army Corps of Engineers.

Why We Are Here

by
J. Ben Rowe[1]

The Florida Game and Fresh Water Fish Commission is composed of five volunteer citizens appointed by the Governor of the State to establish policies and programs in Florida to protect fish and wildlife as well as other natural resources. In our State, we have a daily influx of 800 new residents, all of whom require new infrastructure, roads, schools, housing, and other items necessary for quality life. Additionally, we have millions of tourists visit annually, many of whom enjoy and demand, in addition to our residents, quality fishing, hunting, and other natural experiences.

The Commission, as would any group, prefers to establish policies based upon clear-cut issues with yes and no answers. The use of grass carp in large open systems however is not one of those issues. This fish has been controversial in this State, and many others, for 25 years. I am not sure that even the Commission staff has a consensual opinion on the proper role and use of the grass carp in Florida waters.

Why we are here is simply this. Florida is experiencing serious infestations of submersed weeds, principally hydrilla, and this plant is now present in every watershed in the State. Approximately $6 million/year of public funds are spent annually for chemical, mechanical, and biological control of hydrilla, primarily to maintain fisheries and wildlife habitat, access for boaters, and for flood control purposes. The grass carp has been in the State since 1970, and years of research and experience has been attained by scientists and fisheries biologists.

Locally, Orange and Lochloosa lakes provide 20,000 acres[2] of prime sportfishing, and associated wetlands are used by bald eagles, osprey, otters, snakes, insects, gators, and many other types of critters. A drought, producing low-water levels and extensive hydrilla growth, has reduced lake use by fishermen to nearly zero the past 2 years. In good condition, these lakes have been estimated to produce an economic benefit to this area of over $10 million/year. These lakes have contained various amounts of hydrilla for over 20 years; but in the past 2 years, over $1 million worth of herbicides have been used in these lakes.

It does not take a rocket scientist to figure out that $1 million will purchase many grass carp, and we know that they can provide weed control for 10 to 20 years. It would appear the decision to use the grass carp for aquatic weed control is a simple issue of economics. Unfortunately, this is not so. We have been told that the grass carp is the greatest thing since sliced bread. We have also been told that the grass carp can be an ecological disaster. It is difficult to understand how such opposite opinions can be held by scientists, but the purpose of scientific inquiry is to develop facts on which sound decisions can be formulated. I believe it has been an unwritten policy of the Game and Fish Commission to not stock important fish and wildlife management lakes until we are reasonably assured that environmental degradation will not occur. The public, however, figures that after 25 years of research, how much longer will it take to learn to use this fish? After all, if we can place people on the moon, how difficult can weed control be?

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1 Commissioner, Florida Game and Fresh Water Fish Commission, Gainesville, FL.
2 A table of factors for converting non-SI units of measurement to SI units is presented on page xvi.

I am at this symposium to learn what we are doing in this and other States with regard to the grass carp. Can we manage the fish in our large lakes? Can we remove them if necessary? Will the fish migrate during high water?—Upstream or down? What questions need to be answered, and what facts need to be gathered?

The people in attendance at this meeting can provide many answers to my questions. We greatly appreciate your attendance.

State Policies, Concerns, and Issues Relative to Use of the Grass Carp

Florida Game and Fresh Water Fish Commission Concerns Regarding the Use of Grass Carp

by
Duke Hammond[1]

The Division of Fisheries is concerned with overstocking of triploid grass carp and the introduction of diploid grass carp into the waters of the State. To reduce the chances of either of these events occurring, the Florida Game and Fresh Water Fish Commission requires permits to possess grass carp. All sites larger than 5 acres are inspected by Commission staff prior to permitting. Sites less than 5 acres are permitted following a telephone interview with the applicant. Barriers are required to prevent movement of triploid grass carp out of permitted water bodies. Public water bodies must be approved for stocking by a committee of biologists and administrators from the Commission staff and the Department of Environmental Protection.

The use of grass carp to control noxious aquatic vegetation is a strategy of considerable concern to wildlife managers in the Division of Wildlife. During the early years of development of herbivorous fish as a management tool, the needs of wildlife were often little more than an afterthought. This problem escalated, at least in part, because of the failure of wildlife managers to express these concerns to those researching the use of grass carp.

Wildlife losses resulting from habitat destruction are usually insidious. Impacts to species' population dynamics are often not detected until long after significant damage has occurred. Assessing impacts on wildlife resulting from local anthropogenic manipulations of habitat is greatly complicated for migrating species. They, of course, can be affected by events on the other side of the continent.

Like some fish populations, certain wildlife species can flourish in changing habitat conditions that precede an impending environmental crash. As nutrients enter an aquatic system and become expressed as noxious vegetation, local wildlife use of that system often heightens. This kind of response can result in the misreading of wildlife signs observed over the short-term. If not cautious, observers can become advocates of preserving deteriorating habitat conditions.

Grass carp have proven their effectiveness in certain controlled vegetation management strategies on small (<500 acres) water bodies. It has been repeatedly demonstrated, however, that results are highly variable where grass carp have been used to control vegetation on larger (>500 acres) water bodies. In fact, results are far from consistent on small water bodies. One thing is absolutely certain. Grass carp will consume aquatic vegetation.

That single fact leads us to our greatest concern regarding the use of grass carp. In their eagerness for relief, people plagued by problematic aquatic vegetation are sometimes willing to employ unproven solutions that can lead to other complications. The future, responsible use of grass carp is jeopardized by the frustration of those looking for a quick fix for their vegetation problems.

Hydrilla is usually managed on a crisis basis. Lakes with the most obvious problems are the ones that receive the most public and political attention. The enormous biomass constituted by topped-out hydrilla in large lakes necessitates stocking large numbers of grass carp to effect control. Loss of desirable habitat can result from our inability to remove grass carp once control is effected. Until an efficient

[1] Environmental Administrator, Florida Game and Fresh Water Fish Commission, Tallahassee, FL.

method of recovery is developed, this complication will persist.

A further problem is that sociological and economic rather than biological considerations usually drive the political process regarding vegetation management. If habitat needs of wildlife are not made a high priority, wildlife values will most assuredly decrease as vegetation is managed with other objectives.

Aquatic vegetation houses important links in food webs that sustain wildlife resources that make Florida so aesthetically pleasing. Those resources are of vital economic importance to many Floridians who care a great deal about how well aquatic systems are managed.

It is our hope that wildlife values will be made a high priority in the development of aquatic vegetation management strategies.

Louisiana Grass Carp Policies, Concerns, and Issues

by
Louie V. Richardson[1]

In 1971, the Louisiana Department of Wildlife and Fisheries adopted a State Position banning the importation, transportation, and possession of diploid grass carp, *Ctenopharyngodon idella*. The banning of diploid grass carp was based on the carp's potential to reproduce in our major rivers and a lack of information on the effects of a wild reproducing population on the 350 million acres of important marshland and other inland freshwater areas.

Natural reproduction in Louisiana was first discovered in 1975. Reproduction has now been documented in many of the State's major river systems. In 1983, 23 percent of all fish larvae collected in the Red River was grass carp. Recruitment of larvae to the fingerling stage is apparently severely restricted by hydrological and/or biological events.

Hurricane Andrew in August 1992 created a situation that resulted in a massive, almost total fish kill in the Atchafalaya Basin. An estimated 200 million fish died. The Red River, Ouachita River, and 30 percent of the Mississippi River flow into this great basin swamp. All three river systems have reproducing grass carp populations. No grass carp were noted in the surveys of the massive fish kill in the Atchafalaya basin.

Research has been conducted with diploid grass carp within the State. Four closed system impoundments ranging in size from 30 to 250 acres were stocked with either 15 or 30 grass carp per surface acre in 1974. Results of this project indicated the following:

a. Grass carp effectively controlled coontail, *Ceratophyllum demersum*, and Eurasian watermilfoil, *Myriophyllum spicatum*, in the study lakes.

b. Stocking rates of grass carp should be carefully correlated to the biomass of the target species of aquatic plants.

c. An increase in game fish during the 5-year study period was observed in one study area.

d. Excessive stocking rates of grass carp might have adverse effects on some native fish.

e. Adequate control of dense aquatic vegetation was obtained in 2 to 3 years by stocking fifteen 6-in. fish per surface acre.

f. Grass carp stocked at 6 in. in length attained weights of 15.0 lb in 14 months.

g. Grass carp were readily captured in the study lakes with trammel and gill nets.

h. The use of grass carp for weed control can be useful under certain conditions.

i. Diploid fish should not be introduced into areas that would allow escape into the river systems. Adverse effects could occur, including damage to Louisiana's valuable waterfowl habitat, freshwater wetlands, and estuarine areas.

j. Development of a sterile grass carp could enhance the use of this fish for biological weed control.

Responding to a 1988 Legislative directive, the Department organized a Carp Task Force to review available information on the bighead carp, *Hypophthalmichthys nobilis*, the silver carp, *Hypophthalmichthys molitrix*, and the grass carp. All three species are presently found in Louisiana's waters. Interest in the possible commercial cultural of these fish

[1] Aquatic Plant Research and Control, Louisiana Department of Wildlife and Fisheries, Troga, LA.

stimulated the directive. After a lengthy review, the task force and the Department reported back to the Legislature to retain the ban on the diploid grass carp and recommended banning the importation, transportation, and possession of the silver and bighead carp. They further recommended approval of triploid grass carp for vegetation control in catfish culture ponds under a permitting system to be administered by the Department.

The Department approved a research proposal for three large-scale maintenance operation tests in public waters by Inland Fisheries Division personnel. The research goal was to refine stocking rates of triploid grass carp to achieve control, but not elimination of aquatic vegetation.

Two of the three research lakes have been stocked with the carp to control *Hydrilla verticillata*. The third reservoir is to be stocked in July 1994 to control native vegetation in a city water supply. Stocking rates per acre were determined using total acres infested, species composition, and biomass of each species.

In July 1993, permitting of triploid grass carp began for vegetation control in private and municipal waters.

A major concern in Louisiana is the possibility of overutilization of aquatic plants by triploid grass carp. However, another major concern is invasive exotic plants dominating our water systems and causing a loss of the native plant biodiversity and the organisms dependent upon that diversity. We must carefully weigh the expected results of either extreme and proceed cautiously with the management of these valuable waterways.

Overutilization of aquatic plants by a sterile organism is a reversible situation over time. Loss of biodiversity and establishment of a monospecific population of exotic plants can result in severe restrictions on, or loss of, man's intended uses of the resource. It can also severely disrupt the natural ecological balance of our waterways.

In Louisiana, we are mandated by the Legislature to manage the water resource for multiple-use purposes. A major issue in water resource management is user group conflicts. Conflict resolution is the order of business to manage any of these systems. However, aquatic managers must not be unduly influenced by any one user group or groups to manage for their special interest at the expense of the resource or other user group interest.

A self-defeating inconsistency on the part of misguided individuals seems to result from an inability to discriminate between ecologically useful (nonpersistent and limited effect) and ecologically harmful (persistent and broad-spectrum effects) resource management.

I suggest that our failure to manage living, natural resources will, more often than not, cause rather than prevent continuing declines in desirable environmental quality. To preclude efforts, as suggested by some, to undo man-caused blunders of the past, such as the introduction of harmful exotic species as water hyacinth, *Eichhornia crassipes*, hydrilla, etc., would be an ecological tragedy.

Grass carp alone are not the answer to aquatic vegetation problems. They are a potential tool for aquatic vegetation management. Grass carp could be damaging to the aquatic environment when used incorrectly and/or without safeguards.

South Carolina Grass Carp Policy

by
Steven J. de Kozlowski[1]

Diploid grass carp are illegal in South Carolina; however, triploid grass carp have been legal for use since 1985. To help ensure that only triploid fish are used in the State, the South Carolina Wildlife and Marine Resources Department regulates the importation of grass carp by a permit process. Grass carp permits are issued only to reputable out-of-state suppliers. For each shipment of fish, the supplier indicates the number of fish and destination (stocking site or dealer) on the permit, which accompanies the shipment into the State. All triploid grass carp shipments to South Carolina must be tested by the South Carolina Wildlife and Marine Department at their facility in Columbia to certify sterility. Five percent or 120 fish per shipment (whichever is less) are tested using the Coulter Counter Method. If one diploid is detected, the entire shipment is rejected and escorted out of the State. The permits receive final approval by the Department only after successful ploidy checks.

Hauling trucks and in-state facilities are subject to spot checks by the South Carolina Wildlife and Marine Resources Department to verify ploidy. The production of sterile grass carp in South Carolina is only just beginning. However, at the one production facility currently under development, the Department is requiring special precautions and structures to ensure that diploid brood stock do not escape into public waters. In-state holding facilities (dealers) should also be equipped with escapement barriers. Use of escapement barriers by purchasers is encouraged but not required.

Issues and Concerns

There are no overriding issues or concerns in South Carolina that currently limit the use of grass carp in most waters. However, important considerations for stocking large impoundments in the State include the following:

a. Determining stocking rates that control target species, and minimizing impacts to nontarget aquatic plants.

b. Determining the potential for and impact from movement of sterile grass carp past dam structures to downstream waters.

c. Assessing the impact of grass carp to operations at waterfowl management areas located adjacent to target waters.

[1] Chairman, South Carolina Aquatic Plant Management Council, Columbia, SC.

FLORIDA GAME AND FRESH WATER FISH COMMISSION
APPLICATION TO POSSESS TRIPLOID GRASS CARP
FOR AQUATIC PLANT MANAGEMENT

Please Print!!

1. County________________________ 2. GFC FILE #____________________________
(For Office Use Only)

3. Name of Water Body____________________________ 4. Acreage________________

5. Principal Use of Waters____
(Fishing, boating, retention, fish farming, irrigation, water conveyance, other)

6. Person requesting permit:

a. Name____

Homeowners Association____

Mailing Address____

City______________________State_________________Zip_______________

Home Phone (____)______________ Work Phone (____)_________________

Contact Person or Representative_______________________________________

Site Address____

b. List plants or describe problem_______________________________________

____

****IMPORTANT****

7. Provide a map with directions from a major thoroughfare which will allow the water body to be visited by someone unfamiliar with the area.

8. Mark points on map where water enters or leaves the water body.

9. If barriers are necessary, an additional form must be signed to complete this application. This form will be supplied by Commission personnel after inspection of your site.

10. Recommendations made by Commission employees for stocking grass carp to control or manage aquatic vegetation are made based on research and field observations.

11. Please check one of the following:
____ (a) Lake owned by a single owner.
____ (b) Lake owned and represented by Homeowners Association.
____ (c) Lake owned by private multiple or divided interest owners (persons who own and pay taxes on the bottom of the water body)
____ (d) Lake is public

(Turn page over for Page 2 of application)

Florida Game and Fresh Water Fish Commission Application to Possess Triploid Grass Carp for Aquatic Plant Management (Continued)

PAGE 2/TGC APPLICATION

Complete either a, b, c, or d below based on your response to Section 11.

a. If you have checked 11(a), is your pond twenty acres or less in size, located entirely on your own property with no other owners?
Yes______ No______

b. If you checked 11(b) please indicate name of Homeowners Association and have the president or other authorized representative of the Homeowners Association sign on the line marked Office of Association (below) and Section 12 (below).

Name of Homeowners Association____

Officer of Association____
(Signature) (Title)

Note: **As an Officer of Association, you are signing that the Association at large which owns this water body, wants to stock triploid grass carp.**

c. If you checked 11(c) above, attach a list of all riparian owners, their addresses, and telephone numbers. Do you verify that you have notified all riparian owners and advised them of your intention to apply for a permit to stock triploid grass carp? Yes____ No____. Do all riparian owners consent to the stocking of triploid grass carp in this water body? Yes____ No____.

Note: **The Commission will not issue a permit to stock triploid grass carp until all riparian owners of multiple owner water bodies identified in this application agree to the use of triploid grass carp for aquatic vegetation management and control.**

d. Public waters may be eligible for stocking by the Florida Game and Fresh Water Fish Commission with public funds, if the Commission determines the use of triploid grass carp is a biologically sound management tool and in the best interest of the resource. Determination of public waters eligibility will be made during inspection of the water body and depends if public boat ramps, recreational areas, or picnic ground facilities which allow unrestricted public utilization of the water body are located on or adjacent to the water body. Are you requesting the water body be considered for stocking with public funds? Yes____ No____.

12. I, the undersigned, certify that the information contained within this document is correct and true to the best of my knowledge.

______________________________ ______________________________
Signature of Applicant Date

MAIL COMPLETED APPLICATION TO:

DAVID EGGEMAN
GAME & FRESH WATER FISH COMMISSION
620 SOUTH MERIDIAN STREET
TALLAHASSEE, FLORIDA 32399-1600.

Florida Game and Fresh Water Fish Commission Application to Possess Triploid Grass Carp for Aquatic Plant Management (Concluded)

39-23.088 Regulations Governing Grass Carp.

(1) No person shall take, possess, sell or otherwise transfer, buy or otherwise receive, transport or stock any grass carp without first obtaining a permit therefor from the Commission. Any grass carp inadvertently taken must be immediately returned unharmed to the water.

(2) Permits for grass carp other than triploid grass carp:

Grass carp, other than triploid grass carp, may be possessed only as authorized by permit issued by the Commission for the production of triploid grass carp and subject to the following:

(a) Grass carp, other than triploid grass carp, held outdoors may only be held in a water body that has the lowest point of the top edge of its levee, dike or bank or tank at an elevation of at least one foot above the 100-year flood elevation determined by reference to elevation maps issued by the National Flood Insurance Program, U.S. Department of Housing and Urban Development. Such water body shall have no water discharge. Such water body shall be inaccessible to the public at all times by being securely enclosed by fences with locked gates or by the presence of the permittee or his agents guarding such water body and forbidding public access to such water body.

(b) Grass carp, other than triploid grass carp, held indoors may only be held in a container or tank having no water discharge or having a water discharge through a closed drain system that terminates in a dry-bed, waste-water pond. Such dry-bed, waste-water pond shall not be contiguous to any natural water body nor discharge its waters to any other water body at any time.

(c) Grass carp, other than triploid grass carp, may not be possessed in any number exceeding the number authorized by the permit. Grass carp, other than triploid grass carp, that are produced as a by-product in the production of triploid grass carp shall be destroyed, unless such grass carp that are produced as by-product do not cause the permittee to exceed the number of grass carp, other than triploid grass carp, that the permittee is authorized to possess by permit.

(3) Permits for triploid grass carp:

Triploid grass carp may be possessed, stocked, sold, transferred or transported only as authorized by permit issued by the Commission subject to the following:

(a) Triploid grass carp may be held outdoors only in a water body upon which is placed a structure installed in such a manner as to prevent escape of the triploid grass carp from the water body, or in a water body having a natural configuration that forecloses escape of such triploid grass carp. Such structure or configuration shall be maintained by the permittee as long as triploid grass carp remain in the water body.

(b) No person shall sell or otherwise transfer any triploid grass carp, except as authorized by permit from the Commission in addition to any license required for such transfer or sale by s. 372.65, F.S. No person shall sell or otherwise transfer any triploid grass carp unless the recipient of such grass carp has a permit as provided by this section. A copy of such recipient's permit shall be maintained in the transferor's records for a period of one year following such transfer and made available for inspection upon request of the Commission. Any grass carp sold or otherwise transferred shall be certified as triploid grass carp as provided in s. 39-1.004 prior to such sale or transfer or prior to transporting such grass carp for such sale or transfer. The transferor shall furnish to the Commission a report within 30 days following each calendar quarter indicating the transferor's permit number; the name, address and permit number of each recipient of such triploid grass carp; the date of each sale or other transfer; and the number of triploid grass carp sold or transferred, for each sale or other transfer made during the calendar quarter.

39-23.088 Regulations Governing Grass Carp (Continued)

(c) No person shall transport any triploid grass carp without having a copy of the Commission permit authorizing such transportation accompanying the shipment of triploid grass carp, and without such shipment containing only triploid grass carp, certified as provided in s. 39-1.004, and the certificate shall accompany such shipment of triploid grass carp.

(d) The Commission may deny an application for a permit to stock triploid grass carp in any water body, other than a private pond, if such proposed stocking is inconsistent with the principal or planned use of the water body, the optimum sustained use by the public of the water body's living aquatic resources, or sound biological management principles.

(e) Notwithstanding the provisions of (3)(d) of this section, the Commission may grant an application for a research permit to possess or stock triploid grass carp for legitimate research purposes, subject to the following:

1. The research permit shall expire 12 months from the date of issuance.

2. A detailed research proposal shall accompany the application for the research permit. Such proposal shall state with particularity the research objectives and justifications, research project schedule, research methodology, and safeguards that shall assure that any detrimental effect upon the water body or its living aquatic resources will not be of a permanent or substantial nature.

3. A detailed annual report of research findings, which shall include a description of activities undertaken in the permit period, progress toward research project objectives and proposed activities to be undertaken in the ensuing months, shall be submitted prior to renewal of the research permit. Receipt and approval by the Commission is a condition precedent to renewal of the research permit.

(4) All places where grass carp are possessed shall be subject to inspection by Commission personnel at any time. Such inspection may include obtaining blood samples from grass carp for purposes of ascertaining ploidy.

(5) This rule shall take effect on July 1, 1992.

Specific Authority: Art. IV, Sec. 9, Fla. Const., 369.22, 372.021, F.S. Law Implemented: Art. IV, Sec. 9, Fla. Const., 369.22, 372.26, F.S. History: New -- 6-1-86; Amended -- 7-1-89, 7-1-90, 7-1-92

39-23.088 Regulations Governing Grass Carp (Concluded)

Status of the Grass Carp Program in Texas

by
Philip P. Durocher[1]

In January 1992, the Texas Parks and Wildlife Commission approved the use of certified triploid grass carp for vegetation control in Texas waters. Prior to that time, all grass carp were prohibited in the State. In general, the rules require a permit to possess the triploids, set the criteria for permit approval, allow only licensed Texas fish farmers who hold Exotic Species Permits to sell certified triploids, and continue the total prohibition of diploids in the State.

The recommendation of the Fisheries staff to approve the program was based on several considerations. Overabundant aquatic vegetation was and is a serious problem in small ponds over a large part of Texas. Weed infestations have reached levels that prevent use of some private waters by recreational fishermen. To a lesser degree, some public waters are also being negatively affected.

Under the previous law (total ban), the only practical options available to landowners to control aquatic weeds was either to use herbicides or introduce illegal diploid grass carp. Neither of these options benefit management of public waters. Herbicides, or their byproducts, and illegal diploid grass carp eventually find their way into public waters. Both have the potential to cause long-term damage to aquatic environments.

Because herbicide treatment is expensive, the unlawful importation and introduction of diploids were occurring at an alarming rate. Diploid grass carp were being reported in many of our waters. Since the possibility of large populations of reproducing grass carp was our critical concern, we felt that replacing illegal diploids with sterile triploids would lessen the potential for long-term environmental harm. Permitting triploids gave private pond owners an economic incentive to be legal and stop importing diploids.

Basically, the decision was whether to ignore the problem and assume it did not exist because grass carp were prohibited by law, or take a proactive approach and try to get some measure of control over what was entering the State. We chose the latter. The decision seems to have been a good one since recent surveys indicate for the first time that grass carp are reproducing in one of our rivers.

The rules as implemented did several things. First and foremost, they legalized the introduction of certified triploid grass carp in private waters by permit. Secondly, they set a permit fee of $15 per application plus $2 per individual fish. The funds generated in this "grass carp slush fund" were to be used to administer permit issuance and provide for technical assistance to private pond owners needed as a result of the program.

In the rules themselves, specific permit requirements and criteria are established. For instance, special consideration is given to areas that contain threatened or endangered species and to coastal areas. In most cases, applications from those areas are denied.

The new rules also emphasize that diploid grass carp are still totally prohibited. All triploids must come from out-of-state and must be certified by either the Fish and Wildlife Service or the State before shipping into Texas.

Texas fish farmers' concerns were addressed by allowing permitted landowners to buy triploids only from licensed Texas fish farmers.

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1 Director of Inland Fisheries, Texas Parks and Wildlife Department, Austin, TX.

Texas fish farmers became the middle-men in all transactions, which allowed them to benefit from the activity.

Public waters were handled as a separate issue and were not covered under these new rules. The Department may allow stocking into public waters under another permit system. The same criteria for permit approval are applied to public water requests.

The permitting system implemented reflects a conservative approach. The maximum stocking rate routinely allowed is seven triploids per surface acre. Requests for rates above seven per acre are reviewed on an individual basis. Stockings are normally limited to 5-year frequencies, again unless a review indicates unusual circumstances require more frequent stockings. Under the system, stocking sites requiring less than 100 triploid grass carp are inspected on a random basis. All application sites requiring 100 or more triploids per stocking are inspected before permit approval.

Since the program was implemented in 1992, nearly 2,900 permits have been issued for the use of triploids in private waters. Over 78,000 fish have been stocked. Although this level of activity is substantial, it is considerably less than originally projected.

In summary, the State of Texas allows the permitted use of triploid grass carp in private and public waters. The staff still considers diploids a long-term threat to our environment, particularly in estuarine areas. The decision to allow the permitted use of triploids was an attempt to lessen the illegal importation of diploids, which was occurring.

Description of Grass Carp Policy in Washington State

by
Scott Bonar[1]

Triploid grass carp were legalized for aquatic plant control in Washington State in December 1990. Grass carp stocking is regulated through the Washington State Department of Fish and Wildlife (WDFW), which allows only triploid grass carp to be used. Diploid grass carp are considered deleterious exotics, and their use is not allowed.

Currently, there are no producers of triploid grass carp in Washington, so all fish must be shipped from out-of-state. The following documentation must be completed and approved to use triploid grass carp in Washington (Anonymous 1990):[2] (a) verification from the U.S. Fish and Wildlife Service that the grass carp to be stocked are sterile; (b) certification from the U.S. Fish and Wildlife Service that the fish to be stocked are disease free; (c) a Washington Department of Fish and Wildlife fish-stocking permit; (d) a State environmental policy act checklist (SEPA) describing the site to be stocked and potential impacts; (e) a hydraulic project approval if outlets or inlets to the lake are to be screened; and (f) a list of property owners with land adjacent to the targeted water and their consensus to the proposed stocking.

In addition to the above documentation, other requirements must be met to use grass carp in Washington (Anonymous 1990):[2] The WDFW exotic species policy (POL-4001) must be followed to stock triploid grass carp; only triploid grass carp greater than 8 in. in length may be used; the grass carp must not pose a significant threat to rare native plants or to fish and wildlife; grass carp may only be stocked in naturally closed water systems or those that can be screened with barriers that do not prevent migration of native fish species; and stocking rates for grass carp must be based on the stocking rate model used by the WDFW. The goal for lake plant control in Washington is that a minimum of 25-percent coverage of aquatic plants be maintained.

For private lakes, only the above WDFW requirements must be met. To stock waters having public access, the above requirements must be met along with a lake restoration feasibility assessment meeting the standards of the Washington State Department of Ecology.

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1 Fisheries Biologist, Washington Department of Fish and Wildlife, Olympia, WA.

2 Anonymous. (1990). "POL-5220 Triploid grass carp planting policy," Washington Department of Wildlife, Olympia, WA.

Virginia's Triploid Grass Carp Program

by
Ed Steinkoenig[1]

In Virginia, the use of biological controls for nuisance aquatic vegetation started in 1979 with the importation of 1,600 F_1 hybrid carp (female bighead carp, *Hypopthulmichthys nubilis*, and male grass carp, *Ctenopharyngodon idella*). These fish were produced by J. M. Malone in Lonoke, AR, and stocked into small lakes in the Shenandoah Valley and Piedmont regions of Virginia.

Regulation 23-6 (Virginia Game, Inland Fish and Boat Laws and Regulations) made it illegal to import, without a permit, certain nonindigenous fish species (piranha, walking catfish, and cichlid perch) as far back as 1973. In 1979, grass carp (*C. idella*) were added to this regulation. Prior to 1979, unknown numbers of grass carp (*C. idella*) were imported to Virginia by private pond owners. Based on staff conversations with landowners, it was determined that the Soil Conservation Service was recommending grass carp for aquatic weed control, and landowners were ordering the fish through trade magazines and journals.

In 1979, hydrilla (*hydrilla verticillata*) was misidentified as elodea and planted on the Virginia side of the Potomac River. Hydrilla has become widespread throughout Virginia since then.

In 1984, the Virginia Department of Game and Inland Fisheries (VDGIF) approved the use of the sterile triploid grass carp for weed control. Only triploid grass carp were allowed, and J. M. Malone and Leon Hill were the only approved distributors. Pond owners desiring grass carp were asked to contact one of the two approved distributors, make arrangements for delivery, and contact for a permit. No carp could be delivered without the VDGIF permit. The recommended stocking rate in 1984 was 16 triploid grass carp/acre. This was revised to 16 triploids/vegetated acre in 1987.

In 1989, VDGIF formed a Grass Carp Committee that was comprised of State fisheries biologists. The Committee (a) produced an informative brochure about weed control and the use of grass carp for public distribution, and (b) revised stocking rates based on available literature and staff experience—two rates were established: one for controlling plants and one for eradicating plants, both depending on plant palatability. Current recommended stocking rates (number/vegetated acre) are as follows:

Food Selectivity	Control	Eradicate
Intermediate preference plants	16	32
Preferred plants	8	16

(c) began random ploidy testing of shipments from all distributors in 1990. Every dealer is required to carry 10 extra fish per shipment. Testing was done at Virginia Polytechnic Institute and State University. An effort is made to test 100 fish annually, and they are currently being tested at Virginia Commonwealth University. There are currently 10 approved grass carp distributors making deliveries in Virginia.

VDGIF issues approximately 547 permits requesting 9,000 triploid grass carp annually. However, not everyone issued a permit stocks grass carp. A statewide survey conducted in 1993 suggests that approximately 75 percent of permittees actually stock fish, making the actual stocking estimate 6,750 triploids/year. Currently, there is no charge for the grass

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carp permit. However, effective July 1994, VDGIF will charge $10.00 for each application submitted.

VDGIF does not have a staff large enough to make site inspections of every application.

Introductory Session

Radio-Tagging Studies with Grass Carp in Two Texas Reservoirs

by

Earl W. Chilton II[1] *and Steven M. Poarch*[1]

Introduction

In many areas, particularly Southern States such as Florida and Texas, aquatic plant growth has become a serious problem. Hydrilla (*Hydrilla verticillata*), Eurasian watermilfoil (*Myriophyllum spicatum*), waterhyacinth (*Eichhornia crassipes*), and others disrupt fishing and boating, and decaying plant material adversely affects the taste of public drinking water. Grass carp (*Ctenopharyngodon idella* Val.) have been effective as biological control agents for aquatic macrophytes in many situations (Van Dyke, Leslie, and Nall 1984; Noble et al. 1986; Thompson, Underwood, and Hestand 1988). Chilton and Muoneke (1992) provide an extensive review of grass carp use and biology. Movement is an important factor in any evaluation of grass carp effectiveness. Fish that move extensively over time may be more likely to emigrate to areas where they are unwanted and less likely to remain in targeted areas.

To assess triploid grass carp movement, radio-tracking studies were conducted in two Texas mainstream reservoirs. Few detailed observations of grass carp movement patterns are published, and those usually report on a limited number of fish (Nixon and Miller 1978; Hockin, O'Hara, and Eaton 1989; Bain et al. 1990; Clapp et al. 1993), or provide only short-term tracking information (Nixon and Miller 1978). For example, data from Hockin, O'Hara, and Eaton (1989) and Beyers and Carlson (1993) included only 5 and 14 fish, respectively, whereas Nixon and Miller's (1978) study only lasted 1.5 months because of the battery life of the radio tags used.

Information about diurnal movements is necessary to determine if fish are visiting secondary feeding areas at night. Only two studies, Hockin, O'Hara, and Eaton (1989) and Beyers and Carlson (1993), have reported on grass carp movements over 24 hr, and both were conducted under very limited conditions. Both studies were conducted in canals; without further investigation, it is difficult to assess their usefulness in predicting grass carp diurnal behavior in larger water bodies, where greater movement is possible such as in mainstream reservoirs.

Objectives of this study were to determine the following: (a) magnitude of triploid grass carp movements within each reservoir, (b) distribution of grass carp relative to aquatic vegetation within each reservoir, and (c) diurnal changes in magnitude of grass carp movements.

Study Sites

Lake Texana is formed by the impoundment of the Navidad River (with significant inflows from Sandy and Mustang creeks (Figure 1)) about 50 km east of Victoria, TX, and covers an area of 4,453 ha. Mean depth is 4.7 m, and maximum depth is 13.4 m. The plant community is dominated by hydrilla, coontail (*Ceratophyllum demersum*), and water hyacinth. American lotus (*Nelumbo lutea*), cattail (*Typha latifolia*), smartweed (*Polygonum hydropiperoides*), and pondweed *Potamogeton* spp.) are present and abundant in places. Lake Weatherford is formed by impoundment of the Clear Fork of the Trinity River about 50 km west of the Dallas-Fort Worth area and covers an area of 445 ha.

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Mean depth is 4.6 m, and maximum depth is 12.5 m. Except for pondweed, American lotus, and coontail, found inside cages during an exclosure study, and 1 g of pondweed found outside cages, bulrush (*Scirpus* spp.), is the only macrophyte found in Lake Weatherford (Poarch and Chilton 1992).

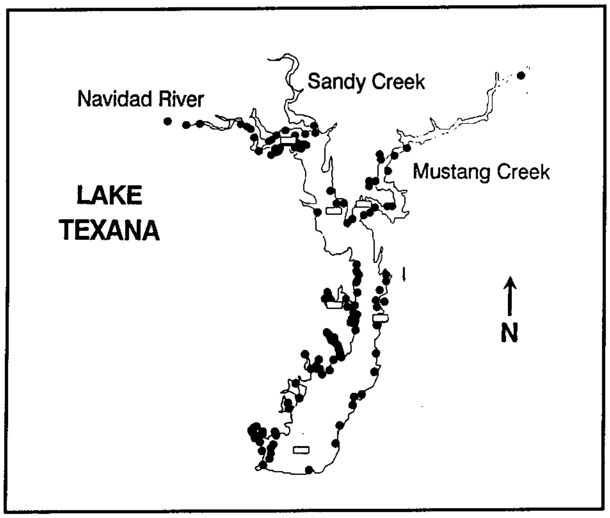

Figure 1. Map of Lake Texana indicating all locations where triploid grass carp stocked in 1990 were found during course of radio-tracking studies (circles) and triploid grass carp stocking sites (rectangles)

Methods

Ninety-five triploid grass carp (mean weight = 1.6 kg, mean total length = 503 mm, size range = 0.5 to 4.7 kg, 369 to 813 mm) were surgically implanted with radio tags equipped with internal loop antennas during June 1990. Tag frequencies ranged from 48 to 50 MHz. Often two tags transmitted on the same frequency, but were distinguished by pulse rate differences (40 or 50 pulses per minute). Fish were divided into groups of 15 or 16 and released at six sites (two stocking sites each in the lower, middle, and upper end of the reservoir) (Figure 1). To locate fish, the entire reservoir was searched by boat, including several miles upstream in the three major tributaries, at 1-week intervals for 3 weeks, 2-week intervals for 8 weeks, and 1-month intervals thereafter.

We conducted 24-hr tracking studies in December 1990 and in February 1991. Ten fish were tracked at 4-hr intervals for 24 hr on each occasion.

In April 1991, 26 additional triploid grass carp (mean weight = 3.75 kg, mean total length = 662 mm, size range = 5.34 to 9.31 kg, 586 to 688 mm) were implanted with tags equipped with external whip antennas, and groups of four or five were released at each of the six original stocking sites. Twelve of the newly stocked fish were tracked at 4-hr intervals for the first 24 hr, and again 48 hr after release. Newly stocked fish were tracked after 1 week, at 2-week intervals for 10 weeks and 1-month intervals thereafter. Another 24-hr tracking study using the same 12 fish was conducted 3 months after release.

In August 1990, 10 triploid grass carp (mean weight = 1.5 kg, mean total length = 495 mm, size range = 1.4 to 1.8 kg, 482 to 506 mm) were implanted with internal loop antenna radio tags and released at one stocking site in Lake Weatherford. Tracking intervals were 1 week for 1 month, 2 weeks for 2 months, and 1-month thereafter. Five fish were tracked at 2-hr intervals for 24 hr in October 1990 and in March 1991.

Fish location was marked on a grid map of each reservoir, and average daily movement (ADM), in $m \bullet d^{-1}$, was calculated for each instance of observed movement. ADM was used to analyze movement patterns through time. Data collected during the first 3 months after release were used to compare differences in movement between 1990 and 1991. Only data from fish located at least four times were used (week 1 movement data for 1991 were not used since there were no week 1 data for 1990). Data were log-transformed to stabilize variances. Core use and home range areas were calculated following Kartalia and Foltz (1994, in preparation).

Results

Lake Texana

Only 69 of the 95 radio-tagged fish originally released in Lake Texana were found during the study. Of those, 6 were located once, 8 were located twice, and 10 were located only three times. Eleven of the forty-five fish located four or more times moved almost exclusively during the first few weeks after release.

Fish released in 1990 were always associated with littoral vegetation, primarily in or near beds of hydrilla and secondarily coontail, followed by American lotus, cattail, smartweed, and pondweed. None were associated with water hyacinth, although it was very abundant in certain areas of the reservoir. Grass carp distribution was skewed toward areas with the most abundant submerged vegetation (Figure 1).

Mean ADM for 1990 was 30.7 m•d^{-1}, and mean ADM for the first 3 months after release in 1990 was 28.4 m•d^{-1}. In general, ADM decreased through time (Figure 2).

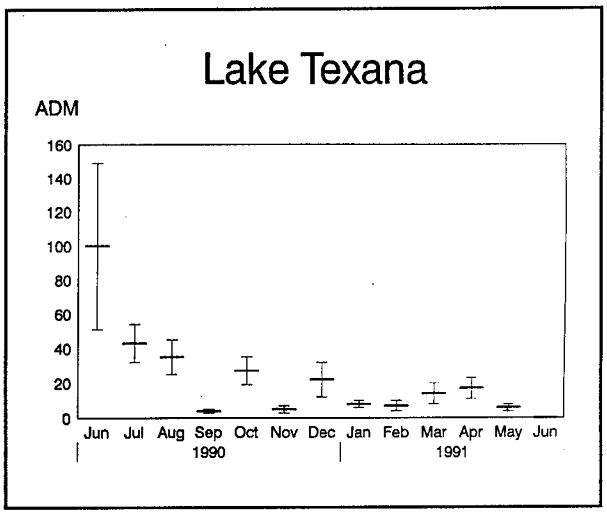

Figure 2. Average daily movement (ADM) (m•d^{-1}) of triploid grass carp stocked in 1990, Lake Texana, Texas. Error bars represent standard errors

Most observed movement of fish released in 1991 occurred within 1 week after release. During the first 3 months, 34.3 percent of movement occurred within the first 2 days after release; 56.4 percent occurred within the first week (movement recorded for the first 2 days after release was not included in this calculation). For the entire study period, 33.1 percent of movement occurred within 2 days, and 47.2 percent occurred within 1 week. ADM was 1,255.4 m•d^{-1} (N = 25, S.E. = 166.4), and 409.3 m•d^{-1} (N = 25, S.E. = 47.9) at 48 hr and 1-week postrelease, respectively. A quiescent period, exhibited by 21 of 26 fish (81 percent) released in 1991, followed initial movement. Mean ADM for the first 3 months after release in 1991 was 116.2 m•d^{-1} (N = 24, S.E. = 15.9).

Fifteen of the twenty-six fish released in 1991 had moved upstream, relative to their initial stocking location, by the last date they were found. In some cases, directionality was difficult to determine.

Average movement during the first 24 hr after release in 1991 was 2.96 km (S.E. = 0.58, range = 0.47 to 7.41 km). However, after 3 months of acclimation, 24-hr tracking studies revealed little diurnal movement for most fish. During the two 24-hr tracking studies with 1990-released fish, 54 percent of our observations indicated zero movement, and 87 percent indicated movement <200 m. Eighty-three percent of movement resulted from fish moving back and forth between stable sites. Fifty-seven percent of observed movement occurred between 0400 and 1200 hr, but was not correlated to water temperature. Similar results were obtained in 1991 (after fish had acclimated) when 78.6 percent of all observations indicated zero movement. Forty-four percent of movement was the result of fish traveling back and forth, and 56 percent of all recorded movement was during 0800 to 1200 hr.

Lake Weatherford

All fish were located at least once. At each location, fish were associated with bulrush, primarily in the upstream portion of the reservoir (Figure 3). As in Lake Texana,

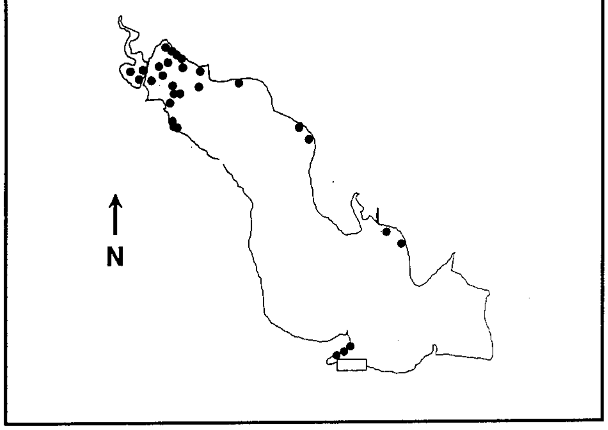

Figure 3. Map of Lake Weatherford indicating stocking site (rectangle) and all locations where triploid grass carp were found during course of radio-tracking studies in 1990 and 1991

movement was most intense during the first week after release; ADM was only about half as great if the first week is treated as an acclimation period (Table 1). Only two fish exhibited movement >200 m after that time, whereas, seven fish began a quiescent period. The period lasted an average of 54.6 days (S.E. = 14.7, range = 6 to 116 days; some fish never moved after the initial period; in those cases, for purposes of calculation, the quiescent period ended at the end of the calendar year).

During 24-hr radio-tracking studies, only one fish moved, and it moved less than 15 m.

Discussion

Our data indicate that upon release, grass carp undergo a period of rapid movement and dispersal, followed by a time of relative inactivity. Nixon and Miller (1978) and Kartalia and Foltz (Personal Communication, Clemson University) reported similar results. A period of inactivity by grass carp, after acclimation, is consistent with their behavior in native habitats. Gorbach and Krykhtin (1988) reported juvenile grass carp may feed for years in the lower reaches of the Amur River. After this period of growth and relatively little long-range movement in the lower portions of the river, they begin an upstream migration to their spawning grounds, traveling as much as 500 km in the first 2 years of the migration. In our study, the fish released in 1991 (mean length 662 mm) were already well within the size range of fish that would have begun their normal upstream migration, and it is not surprising that some exhibited a strong upstream component in their movements.

At least 60 percent of the fish released into Lake Weatherford exhibited upstream movements. However, the data are somewhat difficult to interpret; since virtually all of the plant biomass was found in the upper portions of the reservoir, fish may have been simply going where the food was.

Several fish were found several miles upstream in both the Navidad River and Mustang

Table 1
Average Daily Movement (ADM) of Triploid Grass Carp in Lake Weatherford, Texas

Fish	Overall			Excluding Week 1		
	ADM, $m \cdot d^{-1}$	S.E.	N	ADM, $m \cdot d^{-1}$	S.E.	N
1	64.2	24.6	12	43.1	13.8	11
2	37.6	25.2	12	14.4	10.7	11
3	37.5	22.2	14	16.5	8.1	13
4	0.3	0.3	7	0.3	0.3	7
5	1.7	1.1	14	0.9	0.7	13
6	13.2	10.0	13	3.9	3.9	12
7	53.5	34.4	12	46.3	36.9	11
8	32.9	27.7	13	5.2	1.8	12
9	1.0	1.0	11	0.0	0.0	10
10	117.6	55.6	4	64.8	24.5	3

Note: Overall mean ADM was 36.0 (S.E = 11.5, N = 10), and mean ADM excluding movement during the first week was 19.5 (S.E. = 7.4, N = 10).

Creek, but no fish were located more than 200 m upstream in Sandy Creek. Fish found upstream generally remained in one location over an extended period of time or traveled back down into the main body of Lake Texana. However, movement back into the reservoir was probably precipitated by lack of food in the tributaries where vegetation was sparse. Movement via tributaries could be significant in certain areas of Florida where vegetated lakes are connected by relatively short water courses. After vegetation removal in one lake, it is probable that fish will migrate via such courses if a barrier is not present.

The only instance of significant movement over a 24-hr period was immediately following grass carp release in 1991. All other 24-hr studies indicated minimal diurnal movement, usually during the morning. This result is consistent with data reported by Hockin, O'Hara, and Eaton (1989), who investigated grass carp movement in an unnavigated portion of the Lancaster Canal, Great Britain, and found, although fish moved and fed at all times, activity was greatest around dawn and remained high until noon. As in our study, Hockin, O'Hara, and Eaton (1989) reported diurnal movement was uncorrelated with water temperature; Cassani and Maloney (1991) found that water temperature, 22.2 to 30.3 °C, did not significantly affect grass carp average speed. In general, temperature shifts over 24-hr are probably not sufficient to significantly affect activity.

Acknowledgments

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Grass Carp Reproduction in the Lower Trinity River, Texas

by

Mark A. Webb,[1] Howard S. Elder,[2] and Robert G. Howells[3]

Introduction

Grass carp (*Ctenopharyngodon idella*) were first introduced into research facilities in Alabama and Arkansas in 1963 (Stevenson 1965; Sutton 1977). Their presence in North America and subsequent releases or escapes in open waters prompted concern about possible reproduction in the United States. Stanley (1976) predicted grass carp would spawn in the Mississippi River valley but thought success would be minimal. Stanley, Miley, and Sutton (1978) indicated that large, long rivers or canals with high volumes (>400 m^3/s) and fast flow rates (>0.8 m/s) were needed for successful spawning. Pflieger (1978) suggested that the abundance of grass carp in the middle Mississippi River was too great to result solely from escapement. Indeed, only 2 years later, Conner, Gallager, and Chatry (1980) reported collection of Cyprinid larvae they identified as grass carp in the Mississippi River of Arkansas and Louisiana. Shortly thereafter, Leslie, Van Dyke, and Nall (1982) demonstrated egg hatching could occur at flow rates down to 0.23 m/s, thus dramatically expanding the potential locations and occasions where grass carp might successfully spawn.

Collection of grass carp larvae within the Mississippi Basin continued. Zimpfer, Bryan, and Pennington (1987) found larvae and predicted successful recruitment. Pflieger and Grace (1987) found juvenile grass carp stranded on a desiccated flood plain in Missouri. Brown and Coon (1991) also reported collection of grass carp larvae in the Mississippi River and four of its tributaries in Missouri.

In Texas, the first legal stockings of grass carp occurred in 1981 and 1982 at Lake Conroe, Montgomery and Walker Counties, on the San Jacinto River System in southeastern Texas (Martyn et al. 1986). By 1983, several large grass carp had been collected in the adjacent Trinity River (Trimm et al. 1989). Over the next few years, other grass carp were taken in the San Jacinto River below Lake Conroe, in and below Lake Houston downstream on the San Jacinto River, in Trinity Bay in the mouth of the San Jacinto River, across Trinity Bay in the mouth of the Trinity River, and up the Trinity River itself (Trimm et al. 1989). Continued commercial catches of grass carp in the Trinity River as well as occasional collections of juvenile grass carp in Trinity Bay prompted concern that reproduction may be occurring in the lower Trinity River.

Trimm et al. (1989) dismissed the collection of a <100-mm juvenile grass carp taken near Baytown, TX, as probably an illegally stocked escapee. They further noted ichthyoplankton collections in the Trinity River by Menn and Pitman (1986) in 1984 and 1985 during possible spawning periods failed to collect either grass carp eggs or larvae. However, most grass carp are stocked at sizes substantially larger than 100 mm to avoid predation losses. Further, collections made by Menn and Pitman (1986) were essentially surface tows that could have missed grass carp eggs where turbulence was not sufficient to force the eggs to the surface. Additionally, relatively short duration of incubation and yolksac periods at warmer temperatures, prior to active swimming and net avoidance, suggested sampling

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intervals of more than a few days could fail to detect a significant spawn. The objectives of this study are to determine if grass carp reproduce and recruit to juvenile stages in the lower Trinity River and to characterize the adult population in the lower Trinity River in terms of size, sex, age, fecundity/reproduction, ploidy, and food habits.

Methods

Ichthyoplankton samples were collected from three stations on the lower Trinity River at three sites: Farm to Market Road 787 (FM 787), U.S. Highway 90 (Hwy 90), and Interstate Highway 10 (I-10) (135, 58, and 0 km above Trinity Bay, respectively) (Figure 1) during spring and summer of 1992 and 1993 when surface water temperatures were between 18 and 30 °C. An 0.5-m 560 micron mesh net was used to take three tows at each station once a week. During each tow, the net was allowed to sink to the river bottom, then secured approximately 1.0 m above the river bottom before being retrieved. Samples were taken from the bridge at FM 787 and from a boat approximately 0.5 miles upstream from Hwy 90 and I-19. Additional tows were taken during summer 1993 in backwater areas off the lower Trinity River in an effort to collect larvae that may have moved to calmer water to feed. All samples were preserved using a 10-percent formalin solution and stained with rose bengal to facilitate subsequent specimen removal. Grass carp eggs and larvae were identified at the Texas Parks and Wildlife Department (TPWD) Heart of the Hills research laboratory, with larval identifications confirmed by Dr. Darrel Snyder, University of Colorado Larval Fish Laboratory, Fort Collins.

A 5-kw boat-mounted electrofisher was used to sample for juvenile grass carp off the main river channel during summer 1993. Also, fish kill investigations and sampling records from the Texas Natural Resource Conservation Commission and the TPWD Resource Protection Division were examined for records of juvenile grass carp in the Trinity/San Jacinto Bay area. Adult grass carp were collected by commercial fishermen using 2.4- × 9.1-m hoop nets with 7.6-cm mesh. Between 30 and 50 nets were fished continuously from April 1993 through April 1994 at different locations in side channels of the lower Trinity River above I-10. Each fish was measured to the nearest millimeter total length and weighed to the nearest 10 g. Sex was determined by internal examination and gonads removed and weighed. Age was determined from cross-sectioned otoliths. Blood samples were taken as soon as possible after capture and ploidy determined using a Coulter counter. The alimentary tract was removed from each fish, cut into three equal sections, and preserved in 3-percent formalin prior to examination.

Results

During May and June of 1992, 691 grass carp eggs were collected at FM 787 and Hwy 90; however, no grass carp larvae were collected during 1992. June 1993 sampling yielded 17 grass carp eggs as well as 1,500 grass carp larvae. All grass carp eggs and larvae collected in 1993 were taken on the same day, with eggs collected at FM 787 and Hwy 90 and larvae collected at I-10. Progressive development was seen in eggs taken from the upper and middle stations both years. No eggs or larvae were taken from backwater samples.

Although adult grass carp were seen and collected during backwater electrofishing, no juvenile grass carp were collected. However, fish kill records from TPWD Resource Protection Division show 65 juvenile grass carp between 102 and 178 mm total length (TL) taken from a bayou off the Houston Ship Channel in fall and winter 1993. In addition, one juvenile grass carp (65 mm TL) was collected from the Houston Ship Channel (downstream from the TPWD collection sites) (Figure 1) by the Texas Water Commission in July 1993.

Preliminary results from adult grass carp taken by commercial anglers in the lower Trinity River indicate a length range of 680- to 1,064-mm TL with a mean length of 856 mm,

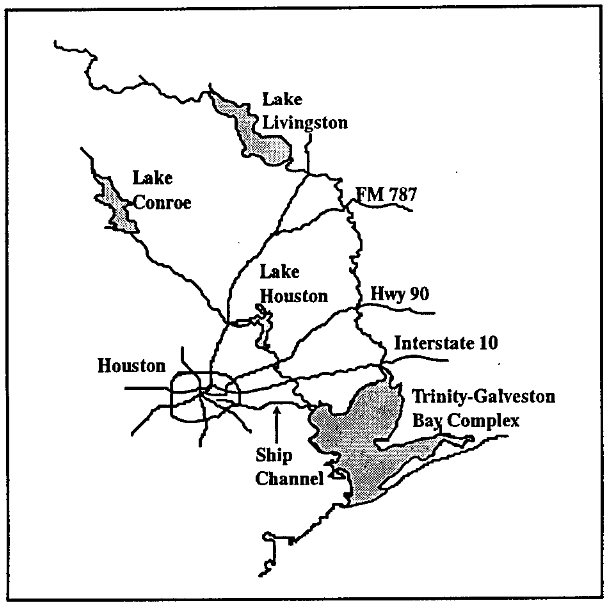

Figure 1. Map of upper Trinity-Galveston Bay Complex including lower Trinity and San Jacinto rivers, Texas

a weight range of 3,400 to 13,630 g with a mean weight of 6,958 g. Sex ratio was 62 percent females and 38 percent males. The age range was from 4 to 9 years with a mean age of 6 years. Peak gonadal weight occurred in June. Eighty-six percent of fish tested were diploid with 14 percent triploid. Many gut samples were empty. For gut samples containing food items, terrestrial detritus (leaves, sticks, etc.) was the most consistent item, with duckweed (*Lemna* spp.) the most common aquatic plant species found. One fish, captured in December 1993, contained large amounts of filamentous algae, the majority being water net (*Hydrodictyon* spp.). Fragments of grass (*Graminae* spp.) and rush (*Scirpus* spp.) were found in small amounts in some samples. Although some invertebrates were found (chironomid larvae and hemiptera), these occurrences were rare and were associated with samples containing large quantities of duckweed.

Discussion

Collection of grass carp eggs, larvae, and small juveniles from the lower Trinity River and Trinity Bay system indicates that successful grass carp recruitment as well as spawning, egg development, and hatching occur in the Trinity River, Texas. Although stocking grass carp has recently been allowed under permit in Texas, legally stocked triploid and illegally stocked diploid grass carp are generally 150- to 200-mm TL or larger at the time of stocking.

Ploidy and age information from the adult grass carp population in the lower Trinity River indicate that at least a portion of the population are escapees from illegal stockings. No saltwater vegetation was found in gut samples, indicating grass carp were feeding on vegetation in the immediate area in which they were captured. More work needs to be done concerning the link between the Trinity River grass carp population and cord grass predation in the San Jacinto Bay.

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Movement of Triploid Grass Carp in Large Florida Lakes

by

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Introduction

Grass carp (*Ctenopharyngodon idella*) are currently being used for aquatic plant control in large, open-lake systems in a number of States, including Alabama and South Carolina, and the ability to accurately predict the fish's vegetation control capabilities will become increasingly important as the use of grass carp in large systems becomes more widespread. Knowledge of localized movement patterns may aid us in determining stocking rates and developing plant control strategies. Nixon and Miller (1978) state that, "... Behavior of grass carp, particularly daily activity, seasonal activity, and distribution is closely allied with their ability to effectively control aquatic vegetation." For example, if we know how large an area that a grass carp will typically cover, and what might cause a fish to move or expand this area, we will be better able to predict the amount of vegetation control provided by that fish. In situations where grass carp are stocked in a system to control a single type or a limited area of aquatic vegetation, movement patterns may determine how well fish achieve this control (Nixon and Miller 1978). Desired plant control may not be achieved if fish move away from a target area of vegetation or never move to that area in the first place.

Vegetation control in an open system may be less than expected if fish emigrate from that system. Ellis (1974), Nixon and Miller (1978), and Hardin and Mesing[3] all provide evidence of grass carp emigration from a stocked system. As of 1978, grass carp had been found 1,690 km from the point of original introduction into the United States, the result of "widely scattered research projects, stockings to solve aquatic weed problems, interstate importation from private hatcheries, and dispersal from stocking sites" (Guillory and Gasaway 1978). Emigration from a given body of water also has the potential to cause undesired impacts on vegetation communities in connected waters (Stanley, Miley, and Sutton 1978), including damage to native aquatic plants and plants important to waterfowl as well as other wildlife food.

Some studies of grass carp movement and behavior have been conducted, but most of this work took place in small (less than 100 ha) lakes (Mitzner 1978), canal (Hockin, O'Hara, and Eaton 1989; Cassani and Maloney 1991), or reservoir (Bain et al. 1990) systems, or for short (less than 30 days) periods of time (Nixon and Miller 1978). Comparatively little research has been conducted to answer questions concerning the movement and long-term behavior of triploid grass carp in large, natural lakes. We conducted the present study to qualitatively and quantitatively describe long-term movement and aquatic plant selection by triploid grass carp in two types of Florida lakes, one open system (several lakes with navigable connections) with little submerged aquatic vegetation and one closed system with extensive submerged aquatic vegetation.

Study Area

Lake Harris (6,688 ha) and Lake Yale (1,636 ha) are part of a large natural chain of lakes in central Florida. Navigable connections exist between Lake Harris and Lake Denham (Helena Run) and between Lake Harris and Lake Eustis (Dead River) (Figure 1). A water

[1] Illinois Natural History Survey, Charleston, IL.
[2] Florida Game and Fresh Water Fish Commission, Eustis, FL.
[3] S. Hardin and C. Mesing, Unpublished Data, Florida Game and Fresh Water Fish Commission.

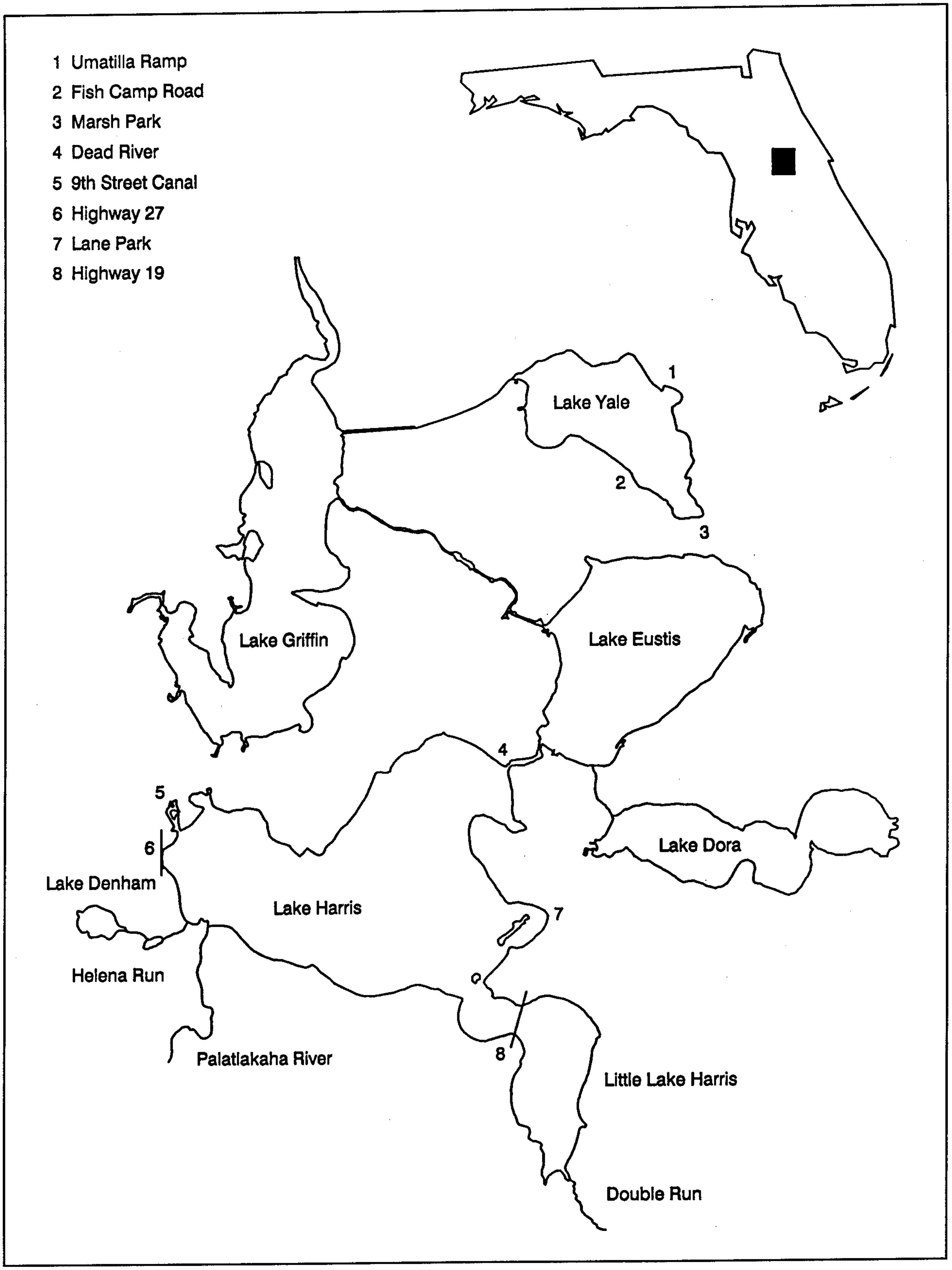

Figure 1. Harris Chain of Lakes, Lake County, Florida

control structure is located in the Palatlakaha River, approximately 2 km upstream from Lake Harris, preventing further movement of fish upstream. Double Run, a historic connection to lakes further up the chain, is now navigable only for approximately 2 km upstream from its entrance into Little Lake Harris. No navigable waterways exist between Lake Yale and other lakes in the system.

Lake Harris is a eutrophic lake, with vegetation limited primarily to a band of emergent plants (predominantly maidencane (*Panicum* spp.), cattails (*Typha* spp.), and arrowhead (*Sagittaria* spp.)) averaging 20 m in width and surrounding the lake to a depth of 2 m. Lake Harris contained significant amounts of hydrilla (*Hydrilla verticillata*) (500 ha) and Illinois pondweed (*Potamogeton illinoensis*) (100 ha) in 1987. A fluridone treatment in that same year was successful in removing the majority of these plants. During the period of this study, submergent plants, including hydrilla and egeria (*Egeria densa*), were limited primarily to Double Run and the Palatlakaha River. These two areas received periodic herbicide treatments during this study, and levels of submergent aquatic plants fluctuated. Lane Park, 9th Street Canal, and the south end of Little Lake Harris contain areas of spatterdock (*Nuphar luteum*) and fragrant water lily (*Nymphea odorata*) from 20 to 100 ha in size.

Lake Yale is a mesotrophic to slightly eutrophic lake with an abundance of rooted aquatic vegetation and occasional algal blooms (Mesing and Wicker 1986). Hydrilla covered approximately 1,200 ha of Lake Yale in 1983. A fluridone treatment in 1984 reduced coverage to a negligible (less than 10 ha) amount. Beginning in 1987, triploid grass carp were stocked at a rate of seven fish/hectare to control hydrilla regrowth in Lake Yale. During the time of this study, hydrilla was found in 150 ha (January 1988) to 500 ha (January 1990) of Lake Yale.

Methods

Radio transmitters from Advanced Telemetry Systems (ATS), Isanti, MN, (Model 5A), were surgically implanted in triploid grass carp following methods outlined in Summerfelt, Hart, and Turner (1972) and Peterson (1975), with minor modifications. Radio transmitters were chosen because signal characteristics are superior in heavy vegetation to those of ultrasonic transmitters (Nixon and Miller 1978). Average weight of the transmitters used was 50 g, and expected battery life was given as 400 to 600 days. The first transmitters used were approximately 9.5 cm long and had 45-cm external antennas. Later in the study, transmitters with coiled internal antennas and mortality circuits that would engage after a transmitter remained motionless for 8 hr were used. The length of these later transmitters was 15 cm. Fish were collected for tagging by electrofishing in lakes previously stocked for vegetation control.

Twenty-two radio-tagged grass carp were released at two locations in Lake Harris between August 1987 and February 1989 (Table 1). These fish were generally released within 8 hr of surgery together with four to five nontransmittered fish, so that transmittered fish would not form a single school. Fish weight ranged from 2.4 to 6.4 kg, and length ranged from 600 to 810 mm. Fourteen fish were radio-tagged and released at three locations in Lake Yale between June 1988 and April 1989 (Table 1). These fish ranged in weight from

Table 1
Radio-Tagged Triploid Grass Carp Stocked in Lake Harris and Lake Yale Between August 1987 and April 1989

Stocking Site	No. of Fish	Days Tracked	No. of Locations
Lake Harris (Aug 1987 - Feb 1989)			
Highway 27	8 (16)	105-546	15-78
Highway 19	4 (6)	38-528	2-66
Lake Yale (June 1968-Apr 1989)			
Umatilla	1 (3)	328	41
Marsh Park	3 (7)	77-432	11-54
Fish Camp Road	3 (4)	264-290	29-33

Note: Number of fish represents those grass carp tracked successfully; total number of radio-tagged fish stocked at a given site is shown in parentheses. Days tracked and number of locations are the range of values for fish at a given site, following a 10-day recovery period. Stocking sites are shown in Figure 1.

2.7 to 6.1 kg and in length from 661 to 827 mm. Since grass carp had been previously stocked in the lake for control of hydrilla, radio-tagged fish in Lake Yale were not released with other, nontransmittered grass carp. We classified a fish successfully tracked if the fish was located for 30 days or longer after a 10-day waiting period following surgery. Nineteen of the thirty-six fish implanted with transmitters were successfully tracked for between 38 and 546 days. In the 17 unsuccessful cases, fish died and were recovered, or transmitters were deposited on the lake bottom as a result of fish dying or expelling transmitters. Transmitters were recovered using a metal detector adapted for aquatic work.

Tracking was conducted by boat approximately once per week on each lake between 0700 and 1630 hr. Average time between locations was 7.2 days on Lake Harris and 8.1 days on Lake Yale. An ATS directional loop antenna (maximum range—800 m) was used for initial long-range location. After initial detection of a radio signal, the boat was maneuvered until a signal could be detected with a short (6-cm) length of coaxial cable or wire. Fish tracked in Lake Harris were located on a gridded map of the lake to the nearest 230-m square grid panel (0.125′ of latitude and longitude), with the center of that panel taken as the actual fish location. Precision of locations for fish tracked on Lake Harris was taken to be one-half the diagonal of a grid square, or 163 m. Later in the study, we obtained a Loran-C recorder, and locations of fish tracked in Lake Yale were recorded to the nearest 0.01′ using this unit. Field experiments using our transmitters showed a maximum reception range of 20 m using a 6-cm coaxial cable as an antenna, indicating that the precision with which we could locate fish in Lake Yale was somewhere between 10 m (Loran-C) and 20 m (worst case scenario with short antenna). Locations of fish in both Lake Harris and Lake Yale were plotted on base maps of each lake constructed from field-corrected Loran-C coordinates. Approximately once every 6 weeks, the entire chain of lakes was surveyed from a fixed-wing aircraft to check for emigration.

Maximum distance moved by a grass carp from the site of stocking and distance moved between consecutive radio locations of an individual grass carp were calculated as straight line movements and are considered minimum distances (Hockin, O'Hara, Eaton 1989; Cassani and Maloney 1991). Based on the above measures of movement, we defined percent maximum distance moved as the maximum distance moved by a fish from its stocking site divided by the distance from that stocking site to the most distant point in the lake, and movement rate as the distance moved by each fish over the course of the study (sum of distances moved between consecutive radio locations) divided by the time that fish was tracked. Differences in maximum distance moved from site of stocking, percent maximum distance moved, and movement rate between groups of fish from each lake were examined using the Mann-Whitney test (Conover 1971).

For fish with greater than 20 locations and tracked for periods longer than 180 days, we calculated a 100-percent minimum convex polygon home-range area using the University of Idaho's Program Home Range (Ackerman et al. 1989). This measure was chosen because we were interested in determining the maximum area that might be used by these fish, and because geometrically based estimates such as the 100-percent minimum convex polygon are more appropriate when subtracting areas (see below) and in cases where data may be autocorrelated (Swihart and Slade 1987). However, this type of geometrically based, rather than statistically based, measure can still be used in comparing home-range areas among categories of organisms (Swihart and Slade 1987).

Areas outside of lake or river boundaries are often included when plotting home ranges of aquatic organisms. To solve this problem, we plotted home ranges on Loran C-corrected base maps of Lakes Harris and Yale using SYGRAPH (Wilkinson 1988), then measured areas outside the lake boundary using a planimeter and subtracted these areas from the calculated home ranges (Dombeck 1979; Mesing

and Wicker 1986). This correction resulted in an average reduction in home-range area of 24 percent from that given by the University of Idaho program. Both uncorrected and corrected 100-percent minimum convex polygon home-range areas are reported here and are hereafter referred to simply as total and corrected home-range areas. Differences in home-range area between groups of fish from each lake were examined using the Mann-Whitney test (Conover 1971). Relationships between sample size and home-range area for fish from each lake and for all fish combined were examined using a one-way analysis of variance (ANOVA) (Conover 1971).

Plant occurrence was measured at each fish location in Lake Yale by taking three "frotus" (a 5-m pole with hooks arranged radially at one end) grab samples at each location and recording plant species collected. The dominant plant at each location was defined as the plant occurring in the majority of these three grabs. These dominant plants were grouped by type, relative to plant management objectives for Lake Yale, as follows: target plant—hydrilla; nontarget plants—Illinois pondweed, southern naiad (*Najas guadalupensis*), coontail (*Ceratophyllum demersum*), cattails, eel grass (*Vallisneria americana*), milfoil (*Myriophyllum* spp.), chara (*Chara* spp.), bladderwort (*Utricularia* spp.), spikerush (*Eleocharis* spp.), spatterdock, fragrant water lily, maidencane, bacopa (*Bacopa* spp.), and nitella (*Nitella* spp.); combination—target and nontarget plants equally dominant; or bare—no plants present. Plant occurrence at fish locations was compared among individual fish using Fisher's exact test (SAS Institute, Inc. 1988). The Bonferroni procedure was used in partitioning P for these multiple comparisons. Plant occurrence in Lake Yale was also measured quarterly along six transects throughout the lake, using the same methods as described for measurement at fish locations. Temporal and seasonal changes in plant occurrence within the lake, in plant use for all fish combined, and differences between lake-wide plant occurrence and plant occurrence at fish locations were evaluated using the likelihood ratio chi-square test (SAS Institute, Inc. 1988). The study period was partitioned into summer (April-September) and winter (October-March) periods, as well as a first (October 1988-June 1989) and second (July 1989-February 1990) half for these analyses. No statistical comparisons were made between plant use by fish and plant occurrence in Lake Harris or between Lake Yale and Lake Harris plant communities since only qualitative visual estimates of plant occurrence (emergent plants, visible submerged plants, or open water) were made at fish locations in Lake Harris.

Results

Movement and behavior in a large, open system

Twelve fish were successfully (based on criteria described above) tracked in Lake Harris. All but one of these fish were tracked for periods longer than 80 days, and six were followed for longer than 300 days. The maximum distance moved from site of stocking for a fish in Lake Harris was 17.1 km (Table 2). Median maximum distance moved from site of stocking for this group was 10.4 km, and median percent maximum distance moved was 80 percent. Only 22 percent of locations were greater than 1.0 km from the previous location (Table 2); 61 percent of these movements

Table 2
Statistics Describing Movement of Triploid Grass Carp Tracked in Lake Harris and Lake Yale

Movement Statistic	Lake Harris	Lake Yale
Number of fish tracked	12	7
Median maximum distance, km	10.4 (1.6-17.1)	3.7 (1.8-5.9)
Median percent maximum distance	80 (13-100)	70 (23-92)
Median movement rate, km/day	155 (43-339)	126 (50-161)
Percent greater than 1.0 km	22 (489)	34 (246)

Note: Maximum distance from site of stocking is a straight line distance; percent maximum distance is defined in the text. Percent greater than 1.0 km is the percentage of individual grass carp movements that were greater than 1.0 km from a fish's previous location. Values in parentheses are ranges for fish in each group except percent greater than 1.0 km, for which the total number of measurements taken is given in parentheses.

greater than 1.0 km occurred during the first half of tracking. Median movement rate for Lake Harris fish was 155 m/day (Table 2).

Home areas for the majority of Lake Harris fish were in either or both of two widely separate areas in Lake Harris—Double Run and the Palatlakaha River (Figure 1). While radio-tagged fish did use some areas out of the main body of the lake, including dead-end and obstructed waterways associated with Lake Harris, they did not move through any of the navigable waterways connecting Lake Harris to other lakes in the Harris Chain. Median total home-range area for Lake Harris fish was 5,337 ha, and ranged from 79 to 8,256 ha (Table 3). Median corrected home-range area was 3,021 ha (range; 61 to 4,954 ha), or approximately 45 percent of the total lake area. There was no significant relationship between sample size (number of locations) and either total or corrected home-range area for fish tracked in Lake Harris (ANOVA, $P = 0.9043$ and 0.8834, respectively).

Table 3
Home Range Statistics for Triploid Grass Carp Tracked in Lake Harris and Lake Yale

Fish Number	Days at Large	No. of Locations	Total Area, ha	Corrected Area, ha
Lake Harris				
121	496	62	6,457	4,468
141	544	68	5,268	1,445
200	546	78	648	362
381	186	31	79	61
401	544	68	8,256	4,951
420	189	21	6,030	3,951
780	528	66	3,477	2,456
811	336	48	5,406	3,586
Median			5,337	3,021
Lake Yale				
190	279	31	160	159
440	328	41	336	310
460	290	29	626	594
610	432	54	306	302
660	264	33	461	445
950	400	50	508	478
Median			398	378

Note: Total area is the 100-percent minimum convex polygon area calculated using the University of Idaho's Program Home Range. Corrected area is the total area with areas outside the lake boundary subtracted using a planimeter. See text for further description.

While shoreline emergent vegetation was abundant in Lake Harris throughout the study, at the time fish were tracked, the Double Run and Palatlakaha River areas provided the only abundant submergent plant forage in the lake. Nine of the twelve fish tracked used at least one of these areas; three fish moved between the two areas on several occasions, a distance of approximately 16 km. Seventy-five percent of these long-distance movements occurred between October and April.

Movement and behavior in a large, closed system

Seven fish were tracked successfully in Lake Yale, and six of these fish were tracked for longer than 250 days. The maximum distance moved from site of stocking for one fish in Lake Yale was 5.9 km (Table 2). The median maximum distance from site of stocking for all Lake Yale fish was 3.7 km, and the median percent maximum distance moved for these fish was 70 percent. Median maximum distance moved from site of stocking for fish in Lake Yale was significantly less than that for fish tracked in Lake Harris (Mann-Whitney, $P = 0.0423$), but percent maximum distance moved by Lake Yale fish was not significantly different from that of Lake Harris fish (Mann-Whitney, $P = 0.4715$). Thirty-four percent of locations in Lake Yale were greater than 1.0 km from the previous location (Table 2); of these, only 39 percent occurred during the first half of tracking. The median rate of movement for triploid grass carp in Lake Yale was 126 m/day; this was not significantly different from that of fish tracked in Lake Harris (Mann-Whitney, $P = 0.1280$).

Median total home-range area for fish tracked in Lake Yale was 398 ha (Table 3), and was significantly smaller than that for fish tracked in Lake Harris (Mann-Whitney, $P = 0.0201$). Median corrected home-range area was 378 ha, about 23 percent of the total area of Lake Yale. The difference between corrected home-range area for fish tracked in Lake Yale and those tracked in Lake Harris, while large, was only marginally significant (Mann-Whitney, $P = 0.0528$). As with fish

tracked in Lake Harris, there was no significant relationship between sample size and either total or corrected home-range area for fish tracked in Lake Yale (ANOVA, P = 0.8183 and 0.8155, respectively).

Plant selection

Distribution of plants in Lake Yale varied significantly from the first half of the study to the second (Likelihood ratio chi-square, P = 0.000, df = 3; Figure 2A), but was not significantly different between winter and summer when data from both halves were combined (Likelihood ratio chi-square, P = 0.816, df = 3; Figure 2B). The temporal change in distribution of plants was the result of increased growth of hydrilla in Lake Yale during the second half of the study period.

Triploid grass carp in Lake Yale were quite similar in their use of plants; only 2 of 21 possible comparisons between individual fish were significant and both involved fish #630 (Table 4). Hydrilla was the dominant plant at 15.4 to 53.3 percent of individual fish locations (Table 4). Comparisons between winter and summer use of plants by all fish combined were close to significant (Likelihood ratio chi-square, P = 0.067, df = 3; Figure 3B). Fish were more likely to use areas with a single dominant plant in summer than in winter, when they made more frequent use of areas with combinations of plants. Comparisons of plant use by fish between the first and second half of the study were not significant (Likelihood ratio chi-square, P = 0.8420, df = 3; Figure 3A).

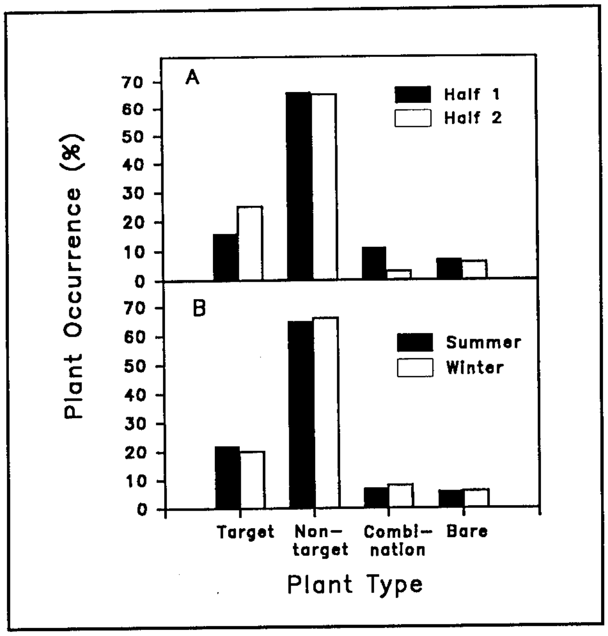

Figure 2. Temporal (A) and seasonal (B) comparisons of lake-wide plant occurrence in Lake Yale, Florida, between October 1988 and February 1990. Plant type groups (target, nontarget, combination, and bare) are based on aquatic plant management objectives; species included in each group are given in text. Temporal half #1 includes the sampling period between October 1988 and May 1989; half #2 includes sampling between June 1989 and February 1990. Summer sampling was conducted from April through September, and winter sampling was conducted from October through March

The distribution of plant types at all fish locations in Lake Yale combined was significantly different from the lake-wide frequency of these same plant types in both summer and winter, during the first and second half of the study, and for all data combined (Figure 4).

Table 4
Statistical Comparisons Involving Overall Plant Use by Individual Fish on Lake Yale

Fish	Target Plant	Nontarget Plant	Combination	Bare	Significant Comparison
190 (28)	39.3	39.3	17.9	3.6	None
440 (38)	21.0	44.7	13.2	21.0	w/630
460 (26)	15.4	30.8	38.5	15.4	None
610 (49)	24.5	26.5	40.8	8.2	None
630 (9)	33.3	0.0	66.7	0.0	w/440,950
660 (30)	53.3	13.3	23.3	10.0	None
950 (46)	30.4	45.6	13.0	10.9	w/630

Note: Values given are percent of fish locations for which the designated plant type was dominant. Description of plant categories is provided in the text. Values in parentheses are total number of locations for each fish. Significant comparisons are those for which plant use was different based on Fisher's exact test ($P < 0.002$). Critical values for Fisher's exact test were adjusted using the Bonferroni procedure to keep overall $P \leq 0.05$. See text for further description.

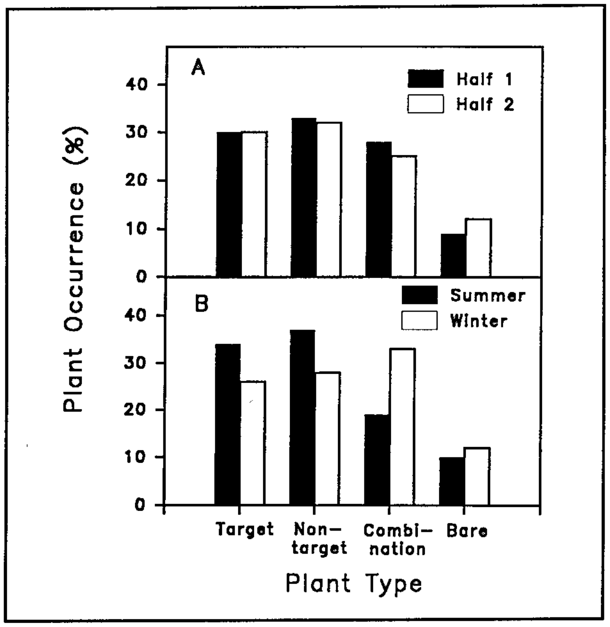

Figure 3. Temporal (A) and seasonal (B) comparisons of plant occurrence at sites used by triploid grass carp tracked in Lake Yale, Florida, between October 1988 and February 1990. Plant type groups and temporal and seasonal sampling periods are as described in Figure 2

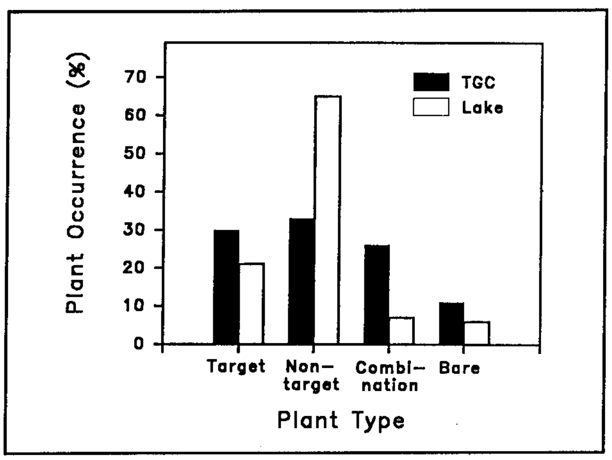

Figure 4. Plant use by triploid grass carp (TGC) tracked in Lake Yale, Florida, as compared with lake-wide distribution of plant types. Data presented are for all sampling periods combined

Hydrilla was the dominant plant at 30 percent of fish locations and only 21 percent of lake-wide sampling locations. Nontarget plants were dominant at 33 percent of fish locations, as compared with 65 percent of lake-wide sampling locations.

Lake Yale triploid grass carp were located in shoreline emergent vegetation, which comprised approximately 5 percent of the lake surface area, on fewer than 10 occasions. Lake Harris fish, on the other hand, were primarily found in association with littoral, emergent vegetation communities, or in canal or riverine environments. These were the only areas in Lake Harris that contained appreciable amounts of submergent vegetation.

Discussion

Distance moved

Distances radio-tagged grass carp moved in this study were within the range of values reported in the literature. Nixon and Miller (1978) observed displacements of 0.2 to 18.3 km, and Martyn et al. (1986) recaptured fin-clipped grass carp up to 9.1 km from the site of release. Grass carp tracked in Florida's St. Johns River were found to move considerable distances upstream and downstream. One fish was located 35 km from point of release.[1]

While statistical comparisons showed, for the most part, few quantitative differences in movement patterns between lakes (other than a difference in maximum distance from site of stocking), we did observe some qualitative differences in movement between the two groups of fish tracked in the present study. These differences may have been related to the distribution of aquatic vegetation in each lake, to seasonal influences, or to variations in water quality within one of the study lakes. In general, movement by Lake Harris fish was intermittent, little or no movement followed by extended movement between separate home areas. Three Lake Harris fish moved on several occasions between two areas approximately 16 km apart; these long-distance movements usually occurred during spring and autumn. Fish in Lake Harris had available to

[1] Personal Communication, F. Cross, Florida Game and Fresh Water Fish Commission.

them only patchy areas of submergent vegetation. When food becomes patchy, animals will tend to limit activity to areas in which food is readily accessible or move intermittently between distant food sources (Pyke, Pulliam, and Charnov 1977).

Stanley, Miley, and Sutton (1978) discuss the importance of temperature in triggering spawning movements, and even though fish used in our study were functionally sterile triploids, spawning instincts may still have played some role in determining movement patterns. Bain et al. (1990) observed different patterns of movement in two groups of triploid grass carp they tracked in Guntersville Reservoir, Alabama, and attributed these differences in movement to differences in fish size and maturity. During the first year of their study, grass carp movements were limited to 6.3 km from the stocking site, but fish tracked during the second year of the study were found up to 72 km from the site of stocking. Somewhat in contrast to our observations, Cassani and Maloney (1991) reported that water temperatures ranging from 22.2 to 30.3 °C did not have a significant effect on grass carp speed, and Nixon and Miller (1978) reported that lower water temperatures decreased rapidity of movement.

Water quality, including dissolved oxygen levels, may also influence movement of grass carp.[1] Previous studies by the Florida Game and Fresh Water Fish Commission have shown that fish will move out of areas with low dissolved oxygen levels.[2] Shireman and Haller reported a decrease in the number of fish locations in some areas of Lake Pearl, Florida, after herbicide treatments.[3] We observed fish to move out of areas of Lake Harris that had been treated with the herbicides endothall and fluridone, but the movements did not occur immediately after the treatments; we did not determine if these fish movements resulted from changes in water quality or changes in the quantity or quality of food available to grass carp in these areas.

Home range

Most studies of grass carp movement have reported some type of home-range tendency on the part of these fish. While Mitzner (1978) indicated that only two of nine fish tracked in Red Haw Lake, Iowa, showed homing to established activity centers, Shireman and Haller reported that 9 of the 10 grass carp they tracked established home areas.[3] Home-range areas were 2.0 to 14.0 ha, and core areas were 0.8 to 6.0 ha. Hockin, O'Hara, and Eaton (1989) described the use of three primary areas by grass carp in a British canal and indicated that fish moved quickly between these areas. Cassani and Maloney (1991) described two general behavior patterns, free-ranging activity and home-area use, and reported that seven of nine fish tracked established use of a home range for some period during their study. Schardt, Jubinsky, and Nall (1982) described home-range use by grass carp as soon as 7 to 10 days following release.

The different results reported above emphasize not only the need for consistency in describing fish behavior but also the importance of the duration of telemetry studies in determining fish behavior patterns. For example, by tracking fish in Lake Harris for over 2 years, we were able to observe use of and movement between two separate areas of the lake by several fish. In general, the longer a fish is tracked, the greater the variety of behaviors likely to be observed and the larger the area it is likely to explore, depending upon whether the requirements (i.e., food and spawning) of that fish can be met in large or limited areas of the system. Fish tracked in Lake Yale had abundant food available to them, and their home-range areas were generally smaller than those of fish in Lake Harris. Lake Harris fish

1 G. Pauley and G. Thomas, Unpublished Data, Washington Cooperative Fisheries Research Unit.
2 Personal Communication, F. Cross, Florida Game and Fresh Water Fish Commission.
3 J. Shireman and W. Haller, Unpublished Data, University of Florida, Gainesville.

were forced to search over a much larger area or to use several different areas since submergent aquatic plants were scarce and patchy.

Home ranges we observed in both lakes were larger than those seen in other studies. One reason for this difference may have been that the lakes we were working on were larger than those in previous studies for which home-range areas of grass carp were reported. As a percentage of lake size, the home-range areas we observed were similar to those reported previously. Home ranges of fish tracked in Lake Harris ranged from 1 to 74 percent of the lake area and of fish tracked in Lake Yale from 10 to 36 percent. Home ranges reported by Shireman and Haller for fish tracked in 24-ha Lake Pearl ranged from 3 to 58 percent of the lake area.[1] Differences between the home-range areas calculated in this study and those reported in other studies may also have resulted from differences in the measures of home range that were used. We reported total and corrected 100-percent minimum convex polygon areas; others have used 95-percent areas or core areas. Our correction (subtracting land areas using a planimeter) resulted in an average reduction from the total home-range area of 38 percent on Lake Harris, but only 4 percent on Lake Yale. This difference was probably due to the increased use of shoreline areas by Lake Harris fish, as compared with those in Lake Yale, as well as their use of multiple areas. Different home-range measures (i.e., statistically based measures) may be more appropriate for studies of different animals or for investigating different research questions. We were interested in determining the maximum area that might be used by triploid grass carp under these conditions, and for this purpose, the corrected 100-percent minimum convex polygon seemed most appropriate.

Plant selection

Knowledge of plant selection by triploid grass carp is of critical importance when considering grass carp for a plant control or management application. Bain et al. (1990) found that grass carp in Guntersville Reservoir, Alabama, chose areas of hydrilla (43 percent at fish locations versus 3 percent lake-wide) and Eurasian watermilfoil (*Myriophyllum* spp.) (20 percent versus 13 percent) in the first year of their study; but in the second year, fish chose primarily Eurasian watermilfoil or mixed plant communities, and 22 percent of the locations were in areas devoid of plants. They attributed these differences in plant choice between years to changes in fish size and experience as well as changes in plant distribution between the first and second year of their study. In the present study, we also observed changes in plant distribution in Lake Yale over the course of the study, but changes in use of plants by triploid grass carp in Lake Yale were seasonal rather than temporal in nature. Fish in Lake Yale were more likely to use areas with a combination of target and nontarget plants in winter and areas with dominant coverage of either target or nontarget plants in summer. Fish in Lake Yale chose areas of hydrilla disproportionately to its abundance in the lake, and we located these fish in shoreline emergent vegetation on fewer than 10 occasions. However, in Lake Harris, where plants were limited to shoreline and lotic areas, fish made almost exclusive use of these areas.

There was some evidence that fish in Lake Yale made use of shoreline vegetation more often than we observed. This evidence consisted of fish collected in these areas by electrofishing, circular clearings in shoreline emergent vegetation, and visual sightings of grass carp in midlake areas, followed by collection of fecal material containing shoreline emergent vegetation at these midlake sites. We also found hydrilla in stomach samples taken from fish captured by electrofishing in shoreline emergent areas between 1900 and 2200 hr, indicating that fish may have been feeding in deeper, open-water areas on hydrilla during daytime and then migrating into nearshore areas at night when water temperatures were favorable and the threat of

[1] J. Shireman and W. Haller, Unpublished Data, University of Florida, Gainesville.

avian predation or of human disturbance (Swanson and Bergersen 1988) was minimal.

Additional tracking between 1800 and 0800 hr would help to answer questions concerning possible inshore nocturnal feeding migrations and use of emergent plants by grass carp. Previous studies of grass carp nighttime behavioral patterns have yielded variable results. Mitzner (1978) indicated that, with the exception of some surfacing behavior observed in evening, nocturnal and diurnal movements were not different. Hockin, O'Hara, and Eaton (1989) reported that grass carp in British canals were most active from 0200 to 1200 hr. Activity lessened somewhat in afternoon and was least in late evening. They indicated that water temperature was probably an important factor in these activity fluctuations. Nixon and Miller (1978) found fish to be more active during daylight hours than at night. However, the increased rest periods between movements at night they reported could have been associated with increased nighttime feeding activity by grass carp. Mitzner (1978) reported that fish were generally sedentary near vegetation and more active in open water areas. In general, activity and feeding are probably inversely related since a herbivorous animal cannot graze effectively when it is highly active.

Management recommendations

Based on the results of this study, we can begin to construct a picture of typical grass carp behavior in large Florida lakes. The fact that median percent maximum distance moved was greater than 50 percent in both of the current study lakes (80 percent in Lake Harris and 70 percent in Lake Yale) indicates that grass carp can potentially cover the majority of a lake from a single stocking site. When plants are present in abundance, grass carp can be expected to feed in an area from 250 to 750 ha in size and remain only locally active. If areas of abundant plants are smaller and/or widely separate, grass carp can be expected to occasionally move longer distances in search of plants and may have multiple home areas. In order to use low levels of fish to control problem plants in large lakes, grass carp stockings should be well-placed around a lake to allow fish to effectively cover areas of target plant abundance, based on the measures of movement described above. While all transmitter fish in this study remained in the open system, it is inevitable that, with a large-scale stocking, some fish will leave if a connection to other lakes is available, especially in a flowing-water situation. Situations in which grass carp have "escaped" from a system have generally been in association with flowing water (Ellis 1974; Nixon and Miller 1978; Hardin and Mesing[1]).

A number of questions remain concerning the use of grass carp for aquatic plant management in large lakes. Grass carp schooling behavior, described by Martyn et al. (1986) and Hockin, O'Hara, and Eaton (1989), could lead to high concentrations of fish in some areas of a lake and low concentrations of fish in other areas. When using grass carp at low rates to control regrowth of problem aquatic plants, a uniform distribution of fish throughout the lake may be more desirable. Increased knowledge of grass carp schooling dynamics may help us to achieve this type of distribution without overstocking a lake.

While studies have shown that grass carp feed preferentially on hydrilla, it is still difficult to predict the amount and timing of aquatic plant control to be expected on target and nontarget species when using this fish. Determination of the dynamics of all aquatic plant populations is paramount in being able to accurately predict grass carp plant control capabilities. We must also realize that because of the dynamic changes associated with aquatic systems, we may never be able to predict plant control success as accurately as we would like. The grass carp has been demonstrated to be an economically feasible plant management tool and should be used in conjunction with the other tools at our disposal to implement effective aquatic plant control programs.

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[1] S. Hardin and C. Mesing, Unpublished Data, Florida Game and Fresh Water Fish Commission.

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An Overview of the Use and Efficacy of Triploid Grass Carp *Ctenopharyngodon idella* as a Biological Control of Aquatic Macrophytes in Oregon and Washington State Lakes[1]

by

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Introduction

Considerable money and effort is expended annually to control excessive plant growth in Oregon and Washington State waters. Acceptable practices for controlling aquatic weeds in Washington State have included chemical treatments, mechanical harvesting, water level reduction, dredging, and bottom screening ("Grass Carp Use in Washington" 1990). With the advent of sterile, triploid grass carp (*Ctenopharyngodon idella*) in 1983, local, State and Federal agencies supported a study conducted by the University of Washington Cooperative Fish and Wildlife Research Unit on the efficacy of grass carp for aquatic plant control in the Pacific Northwest.

At the time this research was initiated, little information was available regarding the efficacy of grass carp in north temperate climates of the United States. The major objectives of this research were as follows:

a. Evaluate current methods for determining separating triploid (sterile) grass carp from diploid (fertile) grass carp.

b. Verify the sterility of triploid grass carp.

c. Evaluate the efficiency of plant control by triploid grass carp in the Pacific Northwest.

d. Determine the effect of triploid grass carp on the ecosystems of several Pacific Northwest lakes, primarily on the water quality and the fish populations.

e. Develop a method for predicting grass carp stocking rates for Pacific Northwest waters.

f. Develop methods for capturing grass carp from overstocked waters.

This research was concluded in 1991, and the following gives a broad overview of our results. Specific information from each of these studies is available through Bonar, Thomas, and Pauley (1987, 1988); Bonar (1990); Bonar et al. (1990); Bonar et al. (1993a, b); Bowers, Pauley, and Thomas (1987); Frodge (1990), Frodge, Thomas, and Pauley (1990, 1991); Pauley et al. (1985, 1987, 1991); Thomas et al. (1989, 1990); and Vecht (1993).

Methods

We evaluated the efficacy of grass carp for aquatic plant control in the Pacific Northwest through field and laboratory studies conducted from 1984-1991 and a compilation of data from literature and questionnaire surveys. Seven Washington lakes and one Oregon lake were stocked with grass carp to evaluate their

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2 Washington Cooperative Fish and Wildlife Research Unit, School of Fisheries, WH-10, University of Washington, Seattle, WA.

effect on aquatic plants, water quality, and fish communities, to examine grass carp growth, and to identify feasible grass carp capture techniques. Experimental lakes were divided by barrier nets to provide a reference area that was not stocked with grass carp and one or two treatment areas where grass carp were stocked. Two external control lakes, one on each side of the Cascade Mountain Range, were monitored throughout the study to provide controls in the event of barrier net failure. Experimental lakes were monitored before and after stocking to provide a before-after-treatment-control (BACI) design. In larger Devils Lake, areas of the plant community were cordoned off from grass carp grazing using nets (exclosures).

Laboratory studies were used to examine grass carp preference and consumption on Pacific Northwest macrophytes and to evaluate techniques to separate triploid and diploid grass carp. Studies were conducted in 4270 l outdoor tanks using both flow-through and recirculating filtering systems. Methods varied by experiment and will be discussed in the descriptions of the individual studies below.

Sterility of the triploid grass carp was not directly verified in these studies, but rather through contact with other researchers from warmer, southern climates where the fish matured more quickly.

Overview Of Various Studies

Genetics and growth of grass carp

Origin of grass carp in the United States. Literature searches and telephone surveys revealed that all grass carp in the United States currently are of Chinese origin.[1] Russian literature revealed that differences in genetics, age of maturity, and spawning times exist between the warmwater-acclimated Chinese stocks and the more northern Russian stocks originating from the Amur River system. However, no directed experimentation has been conducted to identify temperature-related differences in feeding rates between the different stocks.

Separation of triploid and diploid grass carp. The external morphology of diploid and triploid grass carp was studied to determine whether the two types could be separated by differences in external characteristics (Bonar, Thomas, and Pauley 1988). In a comparison of 27 measurements, six scale counts and five fin formulae from two stocks of diploid and triploid fish obtained from commercial producers, analysis of covariance, and discriminant analysis indicated a classification accuracy of only 65 to 85 percent. Clearly, triploid grass carp differentiated by external morphology alone should not be stocked into waters where diploid fish are illegal or would cause management problems.

The accuracy of a Coulter counter, a Coulter counter with channelyzer, and a flow cytometer were also evaluated for separating diploid and triploid grass carp.[2] Using a Coulter counter with a channelyzer resulted in 100-percent correct classification of 167 fish, while using a Coulter counter alone resulted in 1 misclassification and 166 correct classifications. Costs of flow cytometery units ($65,000 - $212,000) and sample processing by flow cytometery ($0.57/sample) were both higher than those for the Coulter counter and the channelyzer ($25,490; $0.36/sample).

Growth rates of triploid grass carp in the Pacific Northwest. Growth rates of triploid grass carp were studied from four Washington lakes (Vecht 1993). Growth was highest in East Pipeline Lake, where grass carp grew from an average of 144 to 6,032 g in approximately 4.3 years. In Keevies Lake,

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1 Unpublished Manuscript, 1994, S. A. Bonar, G. L. Thomas, and G. B. Pauley, "Origin of grass carp in the United States and differences in Asian grass carp stocks."

2 Unpublished Manuscript, 1994, S. A. Bonar, G. L. Thomas, and G. B. Pauley, "Accuracy and cost comparison of flow cytometry and Coulter counter techniques for rapid ploidy analysis of grass carp (*Ctenopharyngodon idella*)."

two size classes of grass carp grew from an average of 144 and 732 to 4,419 g in approximately 4.3 years. In Bull South Lake, carp grew from an average of 144 to 3,701 g in approximately 4.3 years. In Big Chambers Lake, two size classes of grass carp grew from 282 and 223 to 2,363 g in approximately 1.3 years. Fulton condition factors of fish from all lakes ranged from 1.13 to 1.87. The length-weight relationship calculated for grass carp in this study (168 to 855 mm total length) was log weight (g) = -4.53359 + 2.87217 log total length (mm). Scale increments were measured to back-calculate growth at annulus formation. When examining fish growth using regression analysis of back-calculated yearly growth increments, we found that 73 percent of the variation in fish growth was explained by fish age. Triploid grass carp growth rates were as high or higher than growth rates of diploid grass carp from their native range.

Effects of triploid grass carp grazing on Pacific Northwest test lakes

Effects on aquatic macrophytes. In 1986 and 1987, triploid grass carp were stocked into five Washington lakes at rates ranging from 113 to 550 fish/vegetated hectare (4.7 to 15.1 fish/metric ton wet weight of vegetation). Three years after stocking, vegetative cover was reduced in all lakes in amounts ranging from 6 to 67 percent (Bonar 1990). Biomass surveys revealed that waterweed (*Elodea canadensis*) and Sago pondweed (*Potamogeton pectinatus*) were grazed first in eastern Washington lakes followed by coontail (*Ceratophyllum demersum*). Native watermilfoil (*Myriophyllum exalbescens*) exhibited no definite trends. In western Washington, floating-leaved pondweed (*Potamogeton natans*) was reduced or removed, but coexisting watershield (*Brasenia schreberi*) was not grazed appreciably.

Within 3 years of stocking Devils Lake, Oregon, with 27,090 sterile triploid grass carp (180 fish/vegetated hectare; 6.1 fish/metric ton wet weight vegetation), the total volume of vegetation declined by 30 percent, while the total vegetative biomass increased (Bonar et al. 1993a). This biomass increase was due primarily to the expansion of short compact communities of the exotic plant Brazilian waterweed (*Egeria densa*). Comparison of the main lake with exclosure areas protected by net barriers and the results of feeding preference experiments suggested that the increased dominance of *E. densa* was caused by factors other than selective feeding. Furthermore, as the grass carp grew, they seemed to control the expansion of *E. densa* and to maintain species diversity in the plant community. Results from feeding experiments suggested that the size of the grass carp may influence feeding preference and should be considered when determining the palatability of the target plant species. Results indicate that although successful stocking rates will have to be higher for cool northern waters than for warm waters in the Southern United States, the grass carp can still be an economical and effective control of plants in temperate climates.

Effects on water quality. Introduction of sterile triploid grass carp into Keevies Lake and Bull Lake in Washington State resulted in a reduction of surface cover and biomass of the aquatic macrophytes, along with some improvements in water quality (Frodge 1990). In areas previously dominated by floating-leaved species, mean bottom dissolved oxygen (DO) increased from <1 to >3 mg/L. Mean conductivity increased from around 30 to 90 µsiemens and was associated with higher ion concentrations, primarily calcium, which increased from 2 to 4 mg/L. In areas previously dominated by submergent species, surface pH was reduced to <10, surface DO decreased from >20 to 10 to 15 mg/L, and mean bottom DO increased from 2.0 to 4.5 mg/L.

Effects on fish assemblages. Fish samples were taken from several Washington lakes in 1985, 1986, 1988, and 1990 and from Devils Lake, Oregon, in 1986, 1987, and 1988 (Thomas et al. 1989; Marino et al. 1994). Fish were taken by either electroshocking or fyke netting. Age, length, and weight data were collected for several species of fish, including largemouth bass (*Micropterus salmoides*), black crappie (*Pomoxis nigromaculatus*), pumpkinseed sunfish

(*Lepomis gibbosus*), bluegill sunfish (*Lepomis macrochirus*), warmouth (*Lepomis gulosus*), yellow perch (*Perca flavescens*), brown bullhead (*Ictalurus nebulosus*), and rainbow trout (*Oncorhynchus mykiss*). Proportional Stock Density Index (PSD) was used as a method of evaluating the quality of the fishery when appropriate. Numbers of largemouth bass in Keevies Lake declined two orders of magnitude 3 years after stocking grass carp. However, the decline occurred in the control area as well as the two treatment areas, suggesting a natural fluctuation and not a response to grass carp grazing. Growth of largemouth bass increased slightly, while pumpkinseed sunfish and black crappie exhibited no differences. In East Pipeline Lake, there were no changes in either growth or abundance of largemouth bass and pumpkinseed sunfish. The population of largemouth bass in Devils Lake declined from 1986 to 1988 by about 30 percent. However, this decline may have resulted from an increase in sport fishing mortality since increased fishing effort was observed following reductions in plant surface cover.

Effects of dense canopies of aquatic macrophytes on Pacific Northwest lakes

Effects on dissolved oxygen and pH. Dense mats of aquatic macrophytes partitioned the littoral zone of two shallow lakes in Washington State into different habitats of varying water quality (Frodge, Thomas, and Pauley 1990). Horizontal distribution of DO and pH in the lakes was associated with the patchy distribution of aquatic macrophytes in the lake, and the vertical distribution of DO and pH was associated with the location of the canopy in the water column. Elevated concentrations of DO and pH were observed in the canopies of submergent species, while lower DO and pH were observed both in the canopies of floating-leaved species and in the subcanopy water of both growth forms. Vertical mixing of the water column occurred in the open-water areas to a greater extent than within macrophyte beds, resulting in higher subsurface DO concentrations than beneath either submergent or floating plant canopies. Diel changes in DO and pH were associated with the growth form of the macrophytes that formed the plant canopy. Diel pH and DO changes were significant within the canopy of the submergent species that formed in the near-surface waters. Diel changes were not seen in either the surface of floating-leaved macrophytes or in the subcanopy of either growth form. In Keevies Lake, western Washington, dense canopies of overlapping floating leaves of *B. schreberi* reduced summer surface and subcanopy dissolved oxygen to <2 mg/L without significantly changing pH. In canopies of *P. natans*, which has subsurface as well as floating leaves, the surface DO was higher than in adjacent open-water areas, but subcanopy DO was as low as beneath *B. schreberi* canopies. In Bull Lake, in eastern Washington, surface canopies of the submergent species *C. demersum* and *M. exalbescens* regularly had DO concentrations of >30 mg/L and pH > 10. Beneath these canopies of submergent plants, DO of <1 mg/L was common at 0.25 to 0.50 m, and pH was typically 1 to 2 units lower than at the surface.

Sediment phosphorus beneath dense aquatic macrophytes. Dense surface canopies of aquatic macrophytes were associated with significant changes in the physical and chemical water quality of two shallow Pacific Northwest lakes (Frodge, Thomas, and Pauley 1991). Internal loading of phosphorus (P) was observed at the sediment-water interface beneath canopies of *C. demersum* and *M. exalbescens* and in deep open-water areas when DO concentrations were ≤0.4 mg/L. Aerobic release of P was observed at sites with surface covers of green filamentous algae (*Pithophora* sp.) where concentrations of DO were >20 mg/L and pH > 9. An increase in surface P concentrations also was observed in sites dominated by the floating-leaved *B. schreberi* and appeared to be associated with leaf decay within the surface canopy. There was an apparent net loss of phosphorus to the sediments beneath both submergent and floating-leaved canopies when DO concentrations were ≥0.4 mg/L. The removal or reduction of the plant canopies could simultaneously reduce anoxic P release while increasing

aerobic P release. These P cycling mechanisms should be considered in the management of aquatic macrophytes.

Effects on fish. Concentrations of DO measured in dense beds of aquatic macrophytes in two western Washington lakes were below reported lethal limits (≤1 mg/L for largemouth bass and steelhead trout (*Oncorhynchus mykiss*) (Frodge et al. 1994). Temperature, DO, pH, and conductivity were measured synoptically, with all parameters except DO within acceptable ranges for largemouth bass and steelhead survival. The impact of the observed low DO concentrations on endemic fish was examined by field observation, electroshocking, and 72 hr in situ cage bioassay. Largemouth bass were tested in Keevies Lake, a small, shallow, eutrophic lake in western Washington, and steelhead trout were tested in the large, deep, mesotrophic Lake Washington. No mortalities occurred over 72 hr in the open water or surface water in dense patches of Eurasian watermilfoil (*Myriophyllum spicatum*) in Lake Washington where DO concentrations were consistently >4.0 mg/L. No significant mortalities occurred in the open-surface water of Keevies Lake or in the upper 1 m of water in patches of *P. natans* where concentrations of DO were consistently >2 mg/L. Significant fish mortalities occurred at both the surface and at the 1-m depth in dense beds of the floating-leaved macrophyte *B. schreberi* and in the bottom water (<2.0 m) of both floating-leaved and submergent plant species, where DO concentrations were <2 mg/L. When the volume of water in Keevies Lake with DO concentrations > 1 mg/L was recalculated during the period of maximum macrophyte biomass and compared with the total volume of the lake, a 50-percent reduction in available largemouth bass habitat was estimated. The observed lethal DO concentrations and high mortalities in the bioassay tests indicate that at high densities, aquatic macrophytes can have significant detrimental local effects on fish. Habitat degradation by these dense growths of aquatic macrophytes has significant implications for the management of aquatic macrophytes and littoral fisheries.

Use of grass carp for aquatic plant control

World usage of grass carp for aquatic plant control. We used a questionnaire to obtain management and stocking rate information on the grass carp from 28 states in the United States and 11 European countries (Bonar 1990). Most responses indicated grass carp were effective in controlling some plant species but not others. Most respondents reported phytoplankton populations increased following the stocking of grass carp ($P < 0.001$). The number reporting waterfowl population declines was not significant ($P > 0.07$), but this may have been due to the small number of responses to this question. Respondents did not agree on changes in game fish population size, and no consistent changes were observed in either the population size ($P > 0.52$) or body condition ($P > 11$). However, the respondents reported that angling quality improved following stocking ($P < 0.001$), which was primarily attributed to better access to the water. An analysis of grass carp capture techniques revealed angling and nets were more effective than electroshocking, nets only, or nets used in combination with electroshocking. Relationships between stocking rate needed for aquatic plant removal and plant morphology, air temperature of treatment site, and waterway type were developed from stocking rate recommendations provided by the respondents.

Feeding preference of grass carp for Pacific Northwest plants. The importance of 20 Pacific Northwest aquatic macrophyte species as food items for grass carp was evaluated (Bowers, Pauley, and Thomas 1987; Pauley et al. 1991). No significant difference in feeding preference was found between diploid and triploid grass carp. No significant differences were found in relative preference because of either fish density in the holding tanks or to water temperatures of 15, 20, and 25 °C. Feeding rates of triploid grass carp increased between 15 and 20 °C, but not between 20 and 25 °C. In paired two species plant preference trials using diploid and triploid

grass carp, there was a distinct difference in feeding preference between all plant pairs tested except for *P. pectinatus* and curly-leaved pondweed (*Potamogeton crispus*), which were selected equally by grass carp.

The preference for plants by fingerling (mean wt. = 269 g) and larger grass carp (mean wt. = 927 g) did not differ when consumption was tested for three different plant assemblages resembling those occurring in Lake Lawrence and Chambers Lake, Washington, and in a hypothetical nonpreferred plant community made up of species from both lakes (Pauley et al. 1991). However, large and fingerling triploid fish preferences did differ for a plant community consisting of *M. spicatum, E. canadensis, E. densa,* and tapegrass (*Vallisneria americana*) that resembled the community from Devils Lake, Oregon (Bonar et al. 1993a). Fingerling fish found *E. densa* almost unpalatable, while it was a highly preferred plant species when presented to the larger grass carp. When presented with the various multispecies plant communities, grass carp did not remove plants in a preferred species-by-species sequence. Instead they grazed simultaneously on several palatable plants of similar preference before gradually switching to less preferred groups of plants. The relative preference of many plants was dependent upon what other plants were associated with them. The relative preference rank for the 20 aquatic plants tested is shown in Table 1.

Table 1
Preference of Grass Carp for Pacific Northwest Aquatic Macrophytes (Preference declines moving down the table)

Potamogeton crispus = *Potamogeton pectinatus*
Potamogeton zosteriformis
Chara sp. = *Elodea canadensis* = Other thin-leaved *Potamogeton* sp.
EGERIA DENSA (large fish)
Potamogeton praelongus = *Vallisneria americana*
Myriophyllum spicatum
Ceratophyllum demersum
Utricularia vulgaris
Polygonium amphibium
Potamogeton natans
Potamogeton amplifolius
Brasenia schreberi = *Juncus* sp.
EGERIA DENSA (fingerling fish)
Nyphaea sp.
Typha sp.
Nuphar sp.

Relationship between plant chemical composition and their consumption by grass carp. The rate at which triploid grass carp consumed three plant species from different locations was measured and compared with the chemical composition of the plants. Grass carp fed on *E. canadensis* from three lakes at significantly different rates ($P < 0.001$), but did not eat *E. densa* from two lakes at significantly different rates (Bonar et al. 1990). Feeding rate of the grass carp was positively correlated to the concentration of calcium ($r = 0.976$) and lignin ($r = 0.946$), but was negatively correlated to the content of iron ($r = -0.808$), silica ($r = -0.934$), and cellulose ($r = -0.922$). Multiple regression analysis revealed calcium and cellulose content were the most important predictors of consumption rate. These experiments demonstrate that water chemistry and plant chemical composition may affect plant palatability and could in part be responsible for some of the discrepancies in grass carp consumption rate and preference studies.

There were numerous differences in the tissue composition of Eurasian watermilfoil, *M. spicatum*, from two different lakes in Washington State (Pauley et al. 1991). Although calcium and magnesium did not differ between samples from the two lakes, significant differences were observed among samples of silica, cellulose, lignin, phosphorous, iron, zinc, and copper. No significant difference was exhibited between the consumption rates of *M. spicatum* from the two different lakes for either fingerling grass carp or larger carp.

Larger fish (927 ± 148 g; 414 ± 22 mm total length) consumed approximately four to five times as much milfoil as smaller fish (269 ± 18 g; 264 ± 18 mm total length).

Disturbance and time of day effects on consumption rates. The highest consumption rates in tank experiments were in the afternoon and at night, which corresponded to times when water temperatures were the highest (20 to 24 °C; Pauley et al. 1991). These results indicate diurnal heating and cooling could affect grass carp consumption rate in small ponds and coves in lakes. Human disturbance did not affect consumption rates substantially. Feeding rate of grass carp disturbed at random times increased as grass carp acclimated to the disturbance.

Prediction of grass carp stocking rates. An empirical model for predicting grass carp stocking rates was developed from the stocking rate recommendations of 38 grass carp managers worldwide and data from over 100 field trials (Bonar 1990). Stocking rates are predicted by matching the desired plant control objective (eradication or partial control) and the accumulated air temperature units of a site to corresponding values on regression curves that are averages of successful stocking rates used by managers under similar conditions. One version of the model (COVER) predicts stocking rates based on the area of the waterway covered by vegetation (grass carp/vegetated hectare), while the other version (BIOMASS) predicts stocking rates based on the maximum total plant biomass found in the water at the height of the growing season (grass carp/metric ton wet weight of vegetation). The model was fine tuned with the results of the Pacific Northwest grass carp field trials.

Capture of grass carp from vegetated lakes. Seven techniques were evaluated as methods of capture for grass carp in five Washington State lakes containing aquatic vegetation (Bonar et al. 1993b). The capture methods included angling, pop-nets, lift nets, or traps in baited areas; angling in nonbaited areas; heating the water in small areas to attract fish; and herding fish into a concentrated area and removing them with gill nets or seines. Herding fish into a concentrated area was the most effective of the techniques ($P < 0.001$), followed by angling in baited areas. Herding is effective in lakes containing thick vegetation, submerged logs, and other underwater obstructions and may be effective for removing grass carp from overstocked waters or to monitor growth.

Summary

The laboratory and field experiments, as well as an exhaustive questionnaire and literature survey, demonstrated grass carp could be an effective plant management tool in the Pacific Northwest. However, water managers in this area will have to stock grass carp at rates higher than those used in most other regions of the country to achieve adequate plant control. Even when stocked at these higher rates, grass carp are economical when compared with other methods of plant control.

Aquatic plants were not completely eradicated in any of the field sites 4 years after stocking. Monitoring should be conducted for longer periods in Northwest lakes to determine if partial control can be achieved over a longer time period or complete plant control results, a common situation reported by others (Leslie et al. 1987; Colle, Cailteux, and Shireman 1989; Bettoli et al. 1993).

With the levels of aquatic vegetation control we achieved using triploid grass carp, we did not see major adverse impacts on water quality or native fish populations. In fact, water quality improved in the lakes heavily infested with aquatic vegetation, primarily because of an increase in subsurface oxygen concentrations following the removal of aquatic plant canopies.

Triploid grass carp were legalized in Washington State, December 1990. Careful, controlled use of grass carp, strict verification of disease-free and sterile fish, and additional experimentation and monitoring to improve stocking rate predictions and capture techniques

will ensure the grass carp's place as a safe, effective component in region-wide aquatic plant management programs.

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Status of Insect Biocontrol Against Hydrilla and Eurasian Watermilfoil

by

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The development of biocontrol technology for aquatic plant management began in 1959 when the U.S. Army Corps of Engineers (CE) and the U.S. Department of Agriculture (USDA) entered into cooperative research efforts. In the first attempt, classical biological approaches were used. Researchers traveled to the country of origin of the plant and looked for natural enemies.

In general, biological control is the introduction by man of parasites, predators, and/or pathogenic microorganisms to suppress populations of plant or animal pests. Many exotic organisms that are introduced into a new habitat expand rapidly and occupy all available resources. Although populations of these organisms often fluctuate seasonally, they often expand and occupy the maximum-carrying capacity of the system (Figure 1). Biological control agents are used to attempt to reduce these populations to below a problem state (Figure 2).

At the onset of joint CE and USDA research activities 35 years ago, there was skepticism about the ability of biocontrol agents to impact aquatic vegetation, particularly submersed aquatic plant problems. As we have developed potential agents, we realized that these agents are not really tools, but they are resources that are subject to abiotic and biotic factors that need to be managed.

Biological agents can be influenced by climate, weather, geographic barriers, and other abiotic factors. These influences need to be considered when utilizing the agents. In addition, biotic factors, such as predation and various forms of competition, also regulate how potential agents perform.

Biocontrol research on submersed aquatic plants has been conducted using both insects and pathogens to manage hydrilla and Eurasian watermilfoil. Hydrilla is a submersed aquatic plant that clogs waterways and impedes

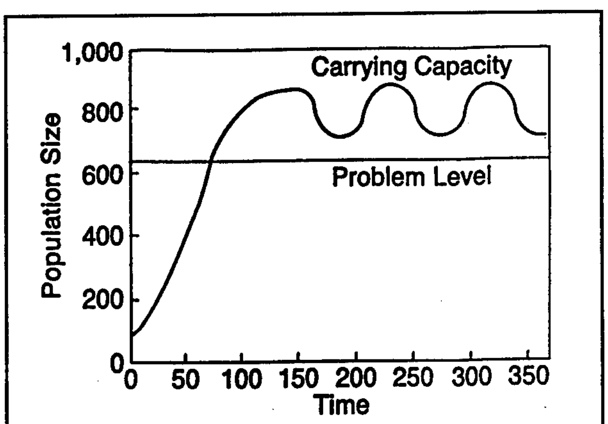

Figure 1. Logistic growth curve of exotic organisms before use of biocontrol agent

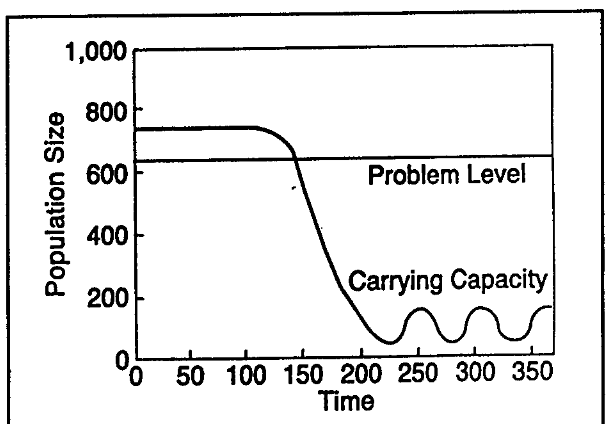

Figure 2. Logistic growth curve of exotic organisms after use of biocontrol agent

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navigation (Schardt and Schmidtz 1989). The plant was introduced into the United States by business, as a plant for fish aquariums. The plant has spread rapidly throughout the southern United States and along the east coast as far north as Delaware. In addition, populations of the plant are found in California.

Hydrilla verticillata L. infested 40,000 to 60,000 acres in Florida alone during the period between 1982 to 1989 (Schardt and Schmitz 1989) and was rapidly expanding into northern regions. Expensive herbicidal control methods were only temporarily effective. Partial treatment with herbicides of an infestation at one lake (Lake Istokpoga) during 1989, for example, cost $1.2 million, an amount equivalent to 20 percent of Florida's total statewide nuisance aquatic plant control budget for that year. Clearly, widespread herbicidal control of hydrilla exceeded the resources of most public agencies, and alternatives were desperately needed. As a result, when the USDA and CE considered collaborative projects for biological control of new target species, *H. verticillata* was positioned as the top priority by both agencies.

Research began in 1980 to determine the area of origin for this problem plant. Surveys were conducted throughout Africa, Australia, and parts of Asia (Balciunas 1982, 1983, 1984, 1985, 1987; Balciunas and Dray 1985). Biocontrol agents were most abundant in India and Australia. In 1984, the USDA established a research facility in Australia with the support of the CE.

The two first insect species selected were from India, a tuber-feeding weevil (*Bagous affinis*) and leaf-mining fly (*Hydrellia pakistanae*). These had originally been discovered in Pakistan during the early 1970s as part of an excess foreign currency (PL-480) project (Baloch and Sana-Ullah 1974; Baloch, Sana-Ullah, and Ghani 1980). Host range tests conducted in the Gainesville, FL, quarantine facility proved that both of these species thrived only on hydrilla (Buckingham 1988a,b, 1989, 1990). Both were released in Florida during 1987 (Center 1989, 1992). *Bagous affinis* failed to establish because of its need for dry conditions, such as a distinct dry season or a prolonged lake drawdown, both atypical occurrences in the Southeast (Bennett and Buckingham 1991; Baloch, Sana-Ullah, and Ghani 1980). Conversely, *H. pakistanae* populations did establish throughout the region.

Faunal surveys were succeeded in Australia by more specific studies. Six candidate Australian species were eventually recognized (Balciunas 1987; Balciunas and Center 1988; Balciunas et al. 1989). Two of these, the leaf-mining fly *Hydrellia balciunasi* and the stem-boring weevil *Bagous hydrillae*, were later imported to the Gainesville, FL, quarantine facility for supplemental testing (Balciunas 1987; Balciunas and Center 1988; Buckingham 1990). Both proved acceptable for field use and were released in 1989 and 1991, respectively. Additionally, four stream-dwelling species of moths were found. These were of interest for their perceived potential to control hydrilla in flowing-water systems, a habitat in which other control measures were particularly ineffective. One species, *Strepsinoma repititalis* (now *Margarosticha repititalis*?), was eliminated from further consideration when it was found that its diet was not restricted to hydrilla (Balciunas, Center, and Dray 1989). A second species, *Nymphula dicentra*, may be conspecific with *Parapoynx diminutalis,* an Asian species already present in the United States, so studies were deferred until its taxonomic status could be ascertained. *Aulacodes siennata* (now *Theila siennata*?) seemed a promising candidate, but the Australian project terminated before evaluations could be completed. A fourth species, *Nymphula eromenalis* (now *Ambia ptolycusalis*?), also had not been evaluated prior to cessation of the project. Dr. Dale Habeck later spent a few months in Australia during 1992 to study these species further. As a result, *A. siennata* was imported into the quarantine facility for host-specificity testing. However, the feeding range exhibited under laboratory conditions proved too broad to risk field release, so it has been dropped from consideration (see Buckingham 1994).

Additional research is continuing in China to study biological control agents that might be useful in temperate regions. As a result, a temperate strain of *H. pakistanae* from northern China was released in 1992. Also, the silver-faced *Hydrellia* species, now named *Hydrellia sarahae*, was imported to United States quarantine, and preliminary testing has begun (see Buckingham 1994). Also, midges similar to the African species presumed to be the causal agent of severe tip damage and stunting of hydrilla plants in Lake Tanganyika (Pemberton 1980) have been found in the Far East (see Buckingham 1994). Additional searches for a stem-boring weevil earlier reported from central China proved fruitless. This bagoine species, unlike the similar *B. hydrillae*, purportedly completes its entire developmental cycle under submersed conditions and would therefore probably be a superior biological control agent. A donaciine beetle was also found associated with hydrilla. Dr. Buckingham observed larvae of these chrysomelid beetles feeding within the root zone of hydrilla beds. The resultant damage caused the plants to appear stressed and generally unhealthy.

Hydrellia pakistanae and *H. balciunasi* are small (about 3 mm) leaf-mining flies that were introduced into the United States from India and Australia, respectively, for the management of hydrilla. *Hydrellia pakistanae* was first released into Lake Patrick, a small south-central Florida lake in 1987 (Table 1). The first releases of *H. balciunasi* occurred in 1989 in a small pond located on the Orangebrook Golf Course in southern Florida and at Lake Patrick in central Florida. Since then, over 4 million *H. pakistanae* and >200,000 *H. balciunasi* have been released in many areas of the southeastern United States.

For *H. pakistanae*, successful establishments have been high. Currently, *H. pakistanae* is apparently permanently established throughout the entire peninsula of Florida from the extreme southern portions to as far north as Gainesville, FL. This indicates that *H. pakistanae* populations are flourishing, and their range is expanding rapidly with little outside help by way of supplementary large-scale releases. In addition, populations of *H. pakistanae* are surviving and expanding in the more northern range of hydrilla. For example, small but persistent populations have been observed in certain sections of Lake Seminole, FL. The most northern established population is found in Muscle Shoals, AL, where high population levels of *H. pakistanae* have occurred since 1992. *Hydrellia pakistanae* is occasionally collected as far west as Sheldon Reservoir near Houston, TX, where it was accidentally released as a contaminant with releases of *H. balciunasi*. *Hydrellia pakistanae* has also been released in considerable numbers in southern Louisiana on Lake Boeuf; recent collections do not reveal the presence of *H. pakistanae*. We are planning to continue to survey the southern Louisiana region for the presence or absence of *H. pakistanae*.

Hydrellia balciunasi is not as widespread as that observed for *H. pakistanae*. While the number and distribution of releases for *H. balciunasi* is considerably less than attempted for *H. pakistanae* (Table 1), releases have been more than adequate in many of the areas. However, the only permanently established population of *H. balciunasi* in the United

Table 1
Total Releases for *Hydrellia pakistanae*, Both the Pakistan and Chinese Strains, and for *H. balciunasi* Since the First Release of *H. pakistanae* in 1987

Species	Number of States	States	Number of Release Sites	Number Released	Date of First Release
H. pakistanae (India strain)	5	FL, GA, AL, LA, TX	27	4,178,223	1987
H. pakistanae (Chinese strain)	3	AL, LA, TX	3	35,800	1993
H. balciunasi	2	FL, TX	9	208,045	1989
Total	5		39	4,422,068	

States occurs in Sheldon Reservoir near Houston, TX, where a consistent but low population has been found since 1992.

Reasons why *H. balciunasi* has not established at more locations are not clear. For example, with numerous release attempts in southern Florida, no established populations can be found. At many Florida sites, establishment appears to be successful at first, but subsequent sampling reveals no *H. balciunasi.* In many of these sites, relative high numbers of *H. pakistanae* are found instead. Such preliminary information may indicate some sort of competitive exclusion between the species, but more information is needed. Similarly, releases of *H. balciunasi* have been accomplished at several sites in the south Texas area; but, to date, no confirmed reports of establishment have been reported except at Sheldon Reservoir. We are currently continuing our release efforts with *H. balciunasi* at several Texas locations, including the Huntsville State Park site and on Choke Canyon Reservoir, where releases were made as recently as fall of 1993 and winter of 1993/1994, respectively.

There is increasing evidence, both qualitative and quantitative, that indicates that the introduced *Hydrellia* sp. cause significant damage to hydrilla infestations. For example, qualitative observations on lakes and ponds since 1987 have indicated that 75 percent had significant declines in the status of the hydrilla infestation after the introduction of *H. pakistanae*. An excellent example is a pond on Orangebrook Golf Course near Fort Lauderdale, FL, where a year after establishment of *H. pakistanae*, a majority of the hydrilla was declining. This decline was correlated with qualitative observations of increases in *H. pakistanae* population levels. Similar declines after the introduction of *H. pakistanae* have been observed at sites in Louisiana and Alabama. For 28 sites that were observed periodically since the initial introductions of *H. pakistanae*, 39 percent had significant reductions in hydrilla after confirmed establishment, and 35 percent had declines even though establishment was weak or not confirmed. In a majority of cases, the introduction of *H. pakistanae* is apparently having significant impact on the hydrilla. However, simple qualitative observations are not sufficient to critically examine the impact of *Hydrellia* on hydrilla infestations. More frequent observations taken more quantitatively are needed for adequately examining the effects of *Hydrellia* on hydrilla infestations.

Myriophyllum spicatum is a submersed aquatic plant native to Eurasia (Godfrey and Wooten 1981). It is believed that the plant was introduced into North America in the early 1800s (Reed 1977); however, other individuals feel it was not introduced until the 1940s (Couch and Nelson 1986). Since its introduction, it has been widely distributed throughout the United States. The plant causes major problems in the northern lakes where extensive monocultures of the vegetation develop and restrict the use of the habitat and outcompete the native vegetation.

Investigations have been conducted of Eurasian watermilfoil declines in North America, and herbivorous insects have been observed associated with these declining watermilfoil populations (Painter and McCabe 1988; MacRae, Winchester, and Ring 1990; Creed and Sheldon 1991). How much impact that these herbivores may have contributed to these declines remains to be determined. Laboratory studies have been conducted which indicate various aquatic insects feed on and damage Eurasian watermilfoil (e.g., Batra 1977; Buckingham and Bennett 1981; Buckingham and Ross 1981; Painter and McCabe 1988; MacRae, Winchester, and Ring 1990; Creed and Sheldon 1991).

Herbivores appear to have a strong effect on watermilfoil buoyancy. The exposure of stem vascular tissue caused by feeding may result in a reduction in buoyancy of the plant. Creed and Sheldon (1991) have observed adult and larval weevils feeding and streams of bubbles emerging from the damaged portion of the stem. This type of impact on a larger scale could result in the collapse of the upper portion of a watermilfoil bed. If herbivores were reducing the flotation capability of the

plant, then removal of considerable amounts of stem and leaf tissue would not be required for the insects to have a strong negative effect on watermilfoil. The reduction in the floating capability of Eurasian watermilfoil may have a secondary impact of dropping plants below the optimal photo zone, reducing the plant ability to photosynthesis, compounding the impact to plant population. The effect becomes even more pronounced if damaged plants can drag down healthy ones. Loss of buoyancy may prove to be one of the major mechanisms of watermilfoil bed destruction by herbivores.

In addition to studies examining the impact of native herbivores on Eurasian watermilfoil, classical biological control research is also being conducted. These research efforts are being based out of the Sino-American Biological Control Laboratory in Beijing, China. Overseas surveys in China began in 1989 to locate insect biocontrol agents of Eurasian watermilfoil. Surveys have been conducted in over 10 provinces in China, and a number of agents have been identified. Two of the most promising candidates, both weevils, are already in United States quarantine facilities in Florida. Presently, inquarantine host specificity studies are being conducted on the *Bagous* sp. and the *Phytobius* sp. of weevils, while studies in China are being conducted on a donaciine leaf beetle larvae that attaches to the roots of Eurasian watermilfoil (Bennett 1994).

All aspects of classical biological control research will continue to examine agents for management of hydrilla and Eurasian watermilfoil. It is believed that the potential exists for effective biocontrol agents to be identified from the native range of these plants because in their native range these plants do not form expensive monocultures of plants that cause problems. It is apparent that some type of control mechanisms is regulating these plant populations.

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Recapture and Removal Session

Removal of Triploid Grass Carp Using Fish Management Bait (FMB)

by

Craig T. Mallison,[1] Rue S. Hestand III,[1] and Boyd Z. Thompson[1]

Introduction

Triploid grass carp *(Ctenopharyngodon idella)* have been used in Florida for aquatic plant management since 1984 (Trent et al. 1992). Although this sterile herbivore is an effective, economical biological plant management tool, it has not been routinely stocked in large public waters because it sometimes removes desirable aquatic vegetation after nuisance target plants are eliminated (Sutton and Vandiver 1986; Leslie et al. 1987; Wiley, Tazik, and Sobaski 1987; Langeland 1990). Once eradication occurs, grass carp may prevent growth of aquatic macrophytes for many years (Van Dyke, Leslie, and Nall 1984; Leslie et al. 1987; Bain et al. 1990; Trent et al. 1992).

Low stocking rates of grass carp may be successful in controlling target vegetation without damaging the native plant community in the short term (Leslie et al. 1987; Phillippy et al. 1989). However, aquatic vegetation growth and the amount of plant control required change over time as a result of many variables, including water levels, weather patterns, nutrient loading, and age-specific feeding preferences of the fish (Osborne and Sassic 1981; Sutton and Vandiver 1986; Wiley, Tazik, and Sobaski 1987). Consequently, long-term selective aquatic plant management using triploid grass carp requires the ability to manipulate the amount of vegetation control (i.e., the number of fish). Supplemental stockings can be used to increase the amount of plant control (Sutton and Vandiver 1986; Trent et al. 1992), but an efficient and practical method of triploid grass carp removal to decrease the amount of plant control has not been developed (Hestand, Thompson, and Phippen 1987; Trent et al. 1992; Bonar et al. 1993). The risks of damaging nontarget vegetation in triploid grass carp aquatic plant management programs will be greatly reduced when a feasible triploid grass carp removal technique is developed.

Fajt and Grizzle (1993) indicated that ingested rotenone is lethal to common carp *(Cyprinus carpio)*. The objective of this study was to determine if triploid grass carp can be removed with a rotenone bait and if so, to develop a methodology to selectively remove significant numbers of triploid grass carp (50 percent or more) for population control.

Materials and Methods

Fish Management Bait (FMB) is a floating fish food containing rotenone that was developed for selective removal of undesirable fish (Prentiss, Inc., Floral Park, NY). Various FMB formulations were produced in attempts to develop an FMB that would be highly preferred by triploid grass carp and were prepared by adding flavor ingredients to the original feed mix prior to extrusion. Flavor ingredients included alfalfa, fish meal, chicken meal, yeast, and a combination of fish meal, chicken meal, and yeast. Trainer pellets for each flavor were produced and contained the same formulation as FMB minus the active ingredient rotenone.

FMB was tested at the Florida Game and Fresh Water Fish Commission's Richloam Fish Hatchery (Sumter County) and seven central Florida lakes. FMB trial procedures included dispensing 250 to 400 g of trainer pellets (small nugget size) once or twice daily from an automatic fish feeder to attract and train

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triploid grass carp. After a feeding response was initiated and stabilized (generally 10 to 25 days), 125 to 400 g of FMB were given to the fish. Uneaten FMB was removed after triploid grass carp had finished feeding. Test sites were inspected for 2 days following FMB application to count and remove dead fish. At Richloam Fish Hatchery, ponds were drained after FMB trials to verify the number of surviving triploid grass carp.

In lakes, the number of triploid grass carp at the feeders was visually estimated by counting fish during feeding. During FMB trials, dying or dead triploid grass carp were marked and returned to the lakes on the first day when possible. On the second day, all dead fish were counted and removed. The number of triploid grass carp killed with FMB was estimated using the Peterson method (Ricker 1975) as follows:

$$N = M * C/R$$

where

N = estimated number of triploid grass carp killed during the FMB trial

M = number of triploid grass carp marked the first day

C = total number of triploid grass carp collected the second day

R = number of marked triploid grass carp collected the second day

Results and Discussion

Richloam Fish Hatchery

Nine FMB trials at Richloam Fish Hatchery were conducted at water temperatures below 23 °C. Many triploid grass carp consumed FMB and became ill (observed porpoising) but subsequently revived. None of the 327 triploid grass carp were removed. In 25 trials at or above 23 °C, 147 of 382 (38 percent) triploid grass carp were removed. During nine initial FMB trials (first test on those fish), 131 of 336 (39 percent) fish were removed (Table 1). As triploid grass carp population density and the number of feeding fish increased, a greater percentage was removed with FMB (Figure 1). This was attributed to more aggressive feeding because of less food per fish. In 16 retrials (tests on survivors of previous tests), 16 of 191 (8 percent) triploid grass carp were removed (Table 2). Previously tested fish avoided FMB, and retrials were much less effective than initial trials. While all flavors of FMB removed at least 20 percent of the triploid grass carp in two trials, the most effective were fish meal and alfalfa.

Lake Monterey

Lake Monterey (0.5 ha, Orange County) was tested with FMB in an attempt to remove the 48 triploid grass carp present in the lake. The most effective trial was the first in which 30 of 48 (63 percent) triploid grass carp were removed with alfalfa FMB (Table 3). Three retrials removed 3 of 18 (17 percent) remaining fish. Overall, 33 of 48 (69 percent) triploid grass carp were removed in the 6-month period from April to October 1993. Nontarget fish found dead included 30 threadfin shad *(Dorosoma petenense)* and seven bluegill *(Lepomis macrochirus)*.

Table 1
Fish Management Bait (FMB) Initial Trials at Richloam Hatchery, 1992-1993

Date	Site	FMB	Number TGC per ha	N / P (%)
8/31/92	Pond 8	Original	1,285	41 / 52 (79)
5/03/93	Pond 1	Fish meal	815	21 / 33 (64)
6/29/93	Pond 1	Alfalfa	692	16 / 28 (57)
6/29/93	Pond 2	Combination	667	14 / 27 (52)
8/10/93	Pond 9	Alfalfa	593	1 / 24 (4)
5/17/93	Pond 1	Yeast	494	2 / 20 (10)
5/17/93	Pond 2	Chicken meal	470	6 / 19 (32)
8/10/93	Pond 15	Alfalfa	259	25 / 68 (37)
9/30/92	Pond 15	Original	247	5 / 65 (8)
1992-93 Total 9 Trials				131 / 336 (39)

Note: Water temperatures were 23-31 °C. Number TGC per ha = number of triploid grass carp per hectare. N = number of triploid grass carp removed, and P = number of triploid grass carp present.

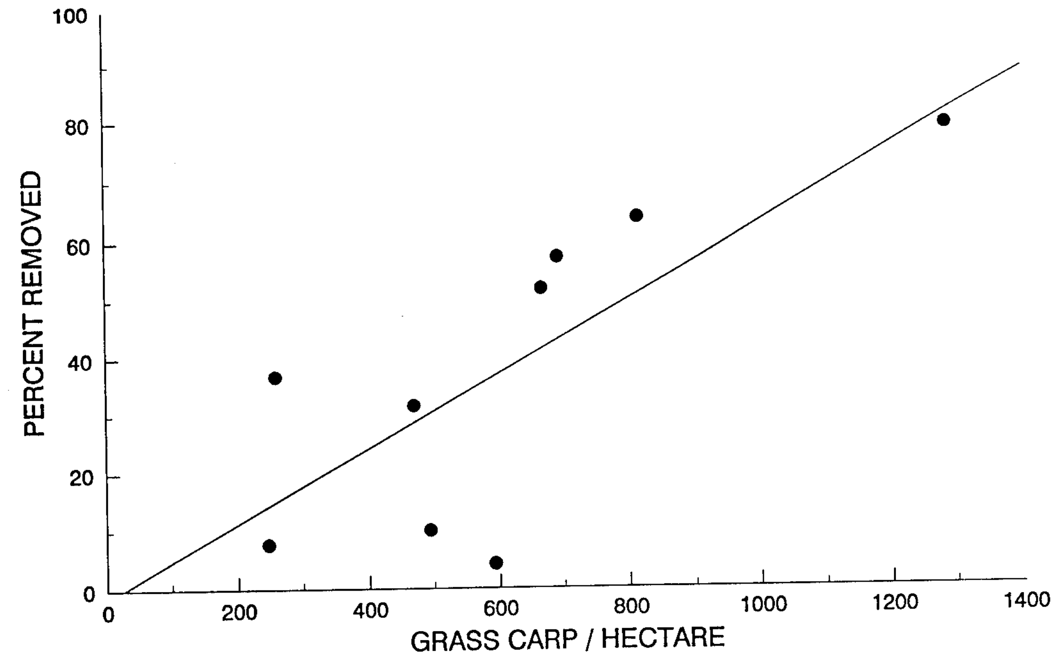

Figure 1. Percentage removal of triploid grass carp with Fish Management Bait was positively correlated with population density (r-square = 0.57)

Table 2
Fish Management Bait (FMB) Retrials at Richloam Hatchery, 1992-1993

Date	Site	FMB	Number TGC per ha	N / P (%)
9/27/93	Pond 9	Alfalfa	568	0 / 23 (0)
7/07/93	Pond A	Alfalfa	536	5 / 52 (10)
9/27/93	Pond A	Alfalfa	484	0 / 47 (0)
10/08/92	Pond A	Original	474	2 / 46 (4)
10/15/92	Pond A	Original	452	1 / 44 (2)
6/07/93	Pond 1	Yeast	445	2 / 18 (11)
5/03/93	Pond A	Fish meal	442	2 / 43 (5)
5/10/93	Pond A	Fish meal	423	1 / 41 (2)
6/08/93	Pond 1	Alfalfa	395	0 / 16 (0)
6/08/93	Pond 2	Alfalfa	321	1 / 13 (8)
6/07/93	Pond 2	Chicken meal	321	0 / 13 (0)
8/10/93	Pond 2	Combination	321	0 / 13 (0)
8/10/93	Pond 1	Alfalfa	297	1 / 12 (8)
5/10/93	Pond 1	Fish meal	297	0 / 12 (0)
9/16/92	Pond 8	Original	272	0 / 11 (0)
9/27/93	Pond 15	Alfalfa	163	1 / 43 (2)
1992-93 Total 16 Trials				16 / 191 (8)

Note: Water temperatures were 23-31 °C. Number TGC per ha = number of triploid grass carp per hectare. N = number of triploid grass carp removed, and P = number of triploid grass carp present. Some fish were tested multiple times, so the total number present is less than the sum of column P.

Table 3
Fish Management Bait (FMB) Trials at Lake Monterey (0.5 ha, Orange County), 1993

Date	FMB	Number TGC per ha	N / P (%)
5/5	Alfalfa	96	30 / 48 (63)
5/20	Alfalfa	36	0 / 18 (0)
7/20	Alfalfa	36	2 / 18 (11)
10/5	Fish meal	32	1 / 16 (6)
1993 Total 4 Trials		96	33 / 48 (69)

Note: Water temperatures were 24-31 °C. Number TGC per ha = number of triploid grass carp per hectare. N = number of triploid grass carp removed, and P = number of triploid grass carp present.

After FMB trials were completed, Lake Monterey was renovated with liquid rotenone (one part per million) to determine the number of triploid grass carp remaining and the species and numbers of other fish present. A total of 2,032 fish weighing 156 kg was collected (Table 4). Fish species present that were not visibly affected by FMB included largemouth bass *(Micropterus salmoides)*, black crappie *(Pomoxis nigromaculatus)*, redear sunfish *(Lepomis microlophus)*, warmouth *(Lepomis gulosus)*, channel catfish *(Ictalurus punctatus)*, brown bullhead *(I. nebulosus)*, blue tilapia *(Sarotherodon milotica)*, lake chubsucker *(Erimyzon sucetta)*, goldfish *(Carassius auratus)*, golden shiner *(Notemigonus chrysoleucas)*, tadpole madtom *(Noturus gyrinus)*, mosquitofish *(Gambusia affinis)*, and swamp darter *(Etheostoma fusiforme)*. Wildlife observed during Lake Monterey trials included black vulture *(Coragyps atratus)*, herring gull *(Larus argentatus)*, anhinga *(Anhinga anhinga)*, mallard duck *(Anas platyrhynchos)*, common egret *(Casmerodius albus)*, common grackle *(Quiscalus quiscula)*, and Florida softshell turtle *(Trionyx ferox)*. No effects of FMB on wildlife were observed.

Table 4
Number and Weight of Fish Removed During Lake Monterey Renovation, November 9, 1993

Fish	Number	Weight, kg	Harvestable[1]	
			Number	Weight, kg
Sport Fish				
Largemouth bass	113	18.3	16	10.2
Black crappie	3	0.6	3	0.6
Redear sunfish	488	36.8	266	33.6
Bluegill	772	23.6	147	14.1
Warmouth	207	2.3	7	0.9
Total Sport Fish	1,583	81.6	439	59.4
Rough Fish				
Brown bullhead	24	9.6	21	9.5
Channel catfish	4	3.8	4	3.8
Grass carp	15	45.2		
Tilapia	3	4.5		
Lake chubsucker	4	2.4		
Goldfish	1	3.2		
Total Rough Fish	51	68.6	25	13.3
Forage Fish				
Golden shiner	6	<0.1		
Madtom	1	<0.1		
Mosquitofish	2	<0.1		
Swamp darter	2	<0.1		
Threadfin shad	387	5.4		
Total Forage Fish	398	5.4		
Grand Total	2,032	155.7	464	72.7

[1] Harvestable lengths are defined here as largemouth bass–300 mm; black crappie–180 mm; redear sunfish, bluegill, and warmouth–160 mm; and channel catfish and brown bullhead–200 mm.

Study lakes

FMB was tested at six lakes ranging from 38 to 152 ha. In 14 initial trials (first application of FMB at that feeder location), 112 triploid grass carp were removed, including 93 of 133 (70 percent) triploid grass carp observed at the feeders (Table 5). In 11 initial trials, 35 triploid grass carp were marked. While 17 (49 percent) were recovered on the following days, recovery rates of marked fish ranged from 25 to 72 percent. The remaining fish were not found because of predation (primarily alligators *(Alligator mississippiensis)*) and large size of the study areas. Based on the Peterson estimates, 145 triploid grass carp (84 percent of the fish observed at the feeders) were killed in 14 initial trials.

During seven retrials (second or third test at that feeder location), FMB removed 7 of 70 (10 percent) fish observed at the feeders (Table 5). Fish fed readily on trainer bait but avoided FMB, indicating that previously tested triploid grass carp detected rotenone in FMB even after waiting 50 days between trials. Futhermore, survivors of initial trials remained at the feeder (as opposed to a new group of fish moving in after some initial fish are removed) and apparently stayed in a home range. Fewer fish were observed at the feeders during retrials than initial trials.

At Live Oak Lake (152 ha, Osceola County), feeders were moved to new locations for the final set of trials in an attempt to attract new groups of triploid grass carp. Since feeding activity was greater at dusk than 9:30-11:00 a.m. (regular testing time), FMB was tested at dusk (7:00 p.m.). This strategy was successful, as FMB removed 27 of 30 (90 percent) fish observed at three feeders. This was more effective than the previous set of trials where FMB removed 14 of 20 (70 percent) fish observed at two feeders (Table 5). Six percent of the estimated number of triploid grass carp in Live Oak Lake was removed with FMB in 6 weeks (five trials). At this rate, one quarter of the population could be removed during the effective annual removal period of 6 months (May to October when water temperatures typically exceed 23 °C in central Florida)

Table 5
Study Lakes Tested with Alfalfa Fish Management Bait (FMB), 1992-93

Lake	Size, ha County	Number TGC per ha	Number Feeder Trials	Number TGC at Feeders	Number TGC Removed (%)	Number TGC Killed (%)
Initial Trials						
Joy[a]	38-Marion	2.6	1	5	5 (100)	5 (100)
Whippoorwill	132-Orange	12.3	3	60	42 (70)	56 (93)
Miona	70-Sumter	2.5	1	0	0	0
Live Oak	152-Osceola	4.9	1	3	0 (0)	0 (0)
Padgett	80-Pasco	15.7	2	-	19 (-)	33 (-)
Mill Dam	60-Marion	5.3	1	15	5 (33)	8 (53)
Live Oak	152-Osceola	4.9	2	20	14 (70)	16 (80)
Live Oak	152-Osceola	4.8	3	30	27 (90)	27 (90)
Subtotal – 6 lakes			14	133[b]	112 (70)[b]	145 (84)[b]
Retrials						
Joy[a]	38-Marion	2.5	1	10	0 (0)	0 (0)
Whippoorwill	132-Orange	11.9	3	30	4 (13)	4 (13)
Whippoorwill	132-Orange	11.9	3	30	3 (10)	3 (10)
Subtotal – 2 lakes			7	70	7 (10)	7 (10)
Total – 6 lakes			21	203[b]	119 (49)[b]	152 (59)b

Note: Water temperatures were 26-31 °C. Number TGC per ha = estimated number of triploid grass carp (TGC) per hectare, number TGC at feeders = number of TGC observed at the feeder(s), Number TGC killed = estimated number of TGC killed (when different from number TGC removed; this is based on second day recovery rates of marked fish - see text), and percent = percent of number TGC at feeders.
[a] Lake Joy was tested with the original FMB formulation (rather than alfalfa FMB).
[b] Lake Padgett trial is omitted from total because number TGC at feeders was not estimated.

if moving feeders consistently attracts new groups of fish.

Nontarget fish found dead during FMB trials at study lakes included 25 golden shiners, 25 Seminole killifish *(Fundulus seminolis)*, and one threadfin shad at Lake Miona (70 ha, Sumter County), five taillight shiners *(Notropis maculatus)* and one warmouth at Lake Whippoorwill (132 ha, Orange County), five taillight shiners at Lake Padgett (80 ha, Pasco County), three bluegill at Lake Joy (38 ha, Marion County), and one largemouth bass at Live Oak Lake. The unusually high bi-kill at Lake Miona was attributed to the absence of triploid grass carp. In this situation, smaller fish had the opportunity to feed much more extensively than when the larger triploid grass carp were present. In the other 20 lake trials, triploid grass carp were present and 15 nontarget fish (less than one per trial) were removed. No effects of FMB on wildlife were observed; species present included otter *(Lutra canadensis)*, alligator, bald eagle *(Haliaeetus leucocephalus)*, osprey *(Pandion haliaetus)*, anhinga, black vulture, turkey vulture *(Cathartes aura)*, great blue heron *(Ardea herodias)*, common egret, snowy egret *(Leucophoyx thula)*, wood stork *(Mycteria americana)*, white ibis *(Eudocimus albus)*, common grackle, common gallinule *(Gallinula chloropus)*, and turtles.

Conclusions

This study showed that triploid grass carp can be removed with a rotenone bait. FMB was effective during most initial trials and removed up to 79 percent of the triploid grass carp present. Alfalfa FMB was the most extensively tested flavor and removed an average 70 percent of the triploid grass carp that were observed at feeders during initial lake trials (mean eight fish per trial). Retrials were ineffective, and each trial removed less than 15 percent of the triploid grass carp present (mean one fish per trial).

In lakes, triploid grass carp remained in home ranges; retrials at each feeder location will likely be ineffective unless success increases after waiting several months between applications. Moving feeders to new locations attracted new groups of fish (equivalent to an initial trial) and may be essential to attract the highest number of vulnerable fish. Using FMB for triploid grass carp population control (removal of at least 50 percent of the population) may be feasible in lakes if moving feeders consistently attracts naive fish and if the majority of the triploid grass carp population can be attracted to feeders. More information about triploid grass carp home ranges is needed to determine the effective area of a single feeder, the distance required to move a feeder out of the home range of tested fish, and the number of trials required to completely treat various size lakes. Future research should also determine optimum feeder location characteristics (depth, cover, distance from shore, etc.) and application times (dawn or dusk), if any.

FMB was selective to triploid grass carp and resulted in minimal bi-kill when triploid grass carp were present. Removal of 51 forage fish (golden shiners, Seminole killifish, and threadfin shad) occurred in one trial where smaller fish had the opportunity to feed thoroughly (triploid grass carp were absent). Uneaten FMB should be removed after triploid grass carp have finished feeding to minimize bi-kill. No effects of FMB on wildlife were observed.

At water temperatures below 23 °C, triploid grass carp were able to recover from FMB. Applications should be made when the water temperature exceeds 23 °C. The efficiency of FMB (percentage removal of triploid grass carp) improved as fish feeding activity increased. Feeding activity was most aggressive when food supply was limited; thus, the amount of trainer pellets and FMB should be minimal (250 to 400 g per feeding time). Two cups (250 g) of FMB was enough to remove up to 30 triploid grass carp at a single feeder.

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Public Angling as a Method of Triploid Grass Carp Removal

by

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Introduction

Triploid grass carp *(Ctenopharyngodon idella)* have been used in Florida for aquatic plant management since 1984 (Trent et al. 1992). Triploid grass carp are an attractive biological control for aquatic plants because they (a) prefer hydrilla *(Hydrilla verticillata)*, which is the most problematic submersed plant in Florida (Hinkle 1986; Langeland 1990), (b) can provide long-term (greater than 3 years) control (Sutton and Vandiver 1986; Wiley, Tazik, and Sobaski 1987; Trent et al. 1992), and (c) are often more economical than alternative methods (Sutton and Vandiver 1986; Wiley, Tazik, and Sobaski 1987). Management has been very successful where eradication of aquatic vegetation is desirable and/or the water body is less than 2 ha in surface area (Trent et al. 1992). However, use of these fish in large lakes has been limited because of potential damage to habitat (Langeland 1990). Triploid grass carp readily consume many native aquatic plant species when hydrilla is not available (Sutton and Vandiver 1986; Langeland 1990), occasionally to the extent of eradication of all submersed and emergent vegetation (Wiley, Tazik, and Sobaski 1987). This can be a problem where aquatic plants serve as important fisheries habitat (Wiley, Tazik, and Sobaski 1987; Porak et al. 1990).

Vegetation growth and the amount of plant control required change over time as a result of changes in water levels, weather patterns, lake and watershed uses, and grass carp feeding behavior and preferences (Osborne and Sassic 1981; Sutton and Vandiver 1986; Wiley, Tazik, and Sobaski 1987). Because of these factors, there is not a stocking formula that applies to all situations (Sutton and Vandiver 1986), and aquatic plant management using triploid grass carp often results in either elimination of nontarget plants as well as targets or insufficient control of target vegetation. This is popularly referred to as the "all or nothing" dilemma of plant management using triploid grass carp.

Selective aquatic plant management using triploid grass carp requires the ability to manipulate the amount of vegetation control (i.e., the number of triploid grass carp). Supplemental stockings can be used to increase the amount of plant control, but an efficient and practical method of triploid grass carp removal to decrease the amount of plant control has not been developed (Hestand, Thompson, and Phippen 1987; Trent et al. 1992; Bonar et al. 1993). Removal of triploid grass carp from overstocked lakes can aid lake restoration projects by reducing grazing pressure on planted aquatic vegetation. In addition, future habitat destruction may be prevented by removing fish before desirable native plants are eradicated. This paper presents a research project with the objective of restoring native vegetation through triploid grass carp removal. The removal technique public angling was explored as a future management option.

Study Area

Lake Mills is a 94-ha, tannin-stained, lake located in Seminole County, Florida. By 1985, infestation of hydrilla had reached 52-percent areal coverage and management was required to restore recreational uses of the lake. The Lake Mills Homeowners Association (LMHOA) implemented an integrated

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aquatic plant management program that consisted of stocking 1,800 triploid grass carp (19 fish/hectare) in January 1986 followed by a fluridone treatment in May 1986. The integrated plant management program resulted in near eradication of aquatic plants. By June 1989, all detectable submersed vegetation was eliminated, and LMHOA requested assistance from the Florida Game and Fresh Water Fish Commission (GFC) with removal of triploid grass carp. The objective was to remove triploid grass carp to allow regrowth of the native plant community, primarily maidencane (*Panicum hemitomon*), eel grass (*Vallisneria americana)*, and stonewort (*Nitella* sp.), to 20-percent areal coverage while maintaining control of hydrilla.

Methods

The GFC has attempted many methods to capture grass carp (Hestand, Thompson, and Phippen 1987; Trent et al. 1992). Angling was determined to be the only practical method for potential removal of triploid grass carp from Lake Mills. Annual permits were issued to 23 interested anglers (possession of triploid grass carp is otherwise illegal in Florida). Four automatic fish feeders that dispensed floating feed pellets were installed to attract triploid grass carp into convenient fishing areas. The most common fishing baits were live worms, dry dog food, bread, and dough balls.

Aquatic vegetation surveys were conducted quarterly to evaluate the response of the plant community to triploid grass carp removal. Surveys consisted of observing emergent plants along the shoreline and sampling submersed plants with a weed hook around the perimeter of the lake. .Total areal coverage was based on occurrence of submersed plants in grab samples and observation of emergent plants.

Results

During 16 months of public fishing, 242 triploid grass carp (3 to 9 kg) were removed by angling (96 percent) and bowfishing (4 percent). This represents 13 percent of the number stocked and 2.6 fish/hectare. Twelve of the twenty-three permitted anglers captured at least one fish, five people caught 10 or more, and three people combined for 170 removals (70 percent of total). More than half of the triploid grass carp removed (126 fish or 53 percent) were captured during the first 2 months of the program. The decreasing catch rates reflect a high initial effort by fishermen in addition to declining catch per unit effort that may have resulted from fewer triploid grass carp present as well as fish presumably becoming wary of boats and anglers. Although effort was not measured by participants, they reported that the fish became much harder to catch as time progressed.

At the beginning of the removal program, there was 1-percent coverage of emergent vegetation and no detectable submersed vegetation. By January 1990 (7 months into the program), 169 triploid grass carp had been removed, and plants began to re-establish in canals and scattered locations along the shoreline. In 1991, total vegetation coverage was 6 percent (2-percent emergent plants, 4-percent submersed plants) and dominated by filamentous algae *(Lyngbya* sp. and *Spirogyra* sp.) and water moss *(Fontinalis* sp.).

Discussion

The number of triploid grass carp in Lake Mills was estimated using a mortality rate of 15 percent for the first year after stocking and 5 percent for the following years (Hestand, Thompson, and Clapp 1990). Adding mortality from the public angling program, the estimated number of triploid grass carp in Lake Mills in January 1992 was 962 (Figure 1). Without a removal program, the estimated number would have been 1,184. Based on this, the 16-month program may have accelerated mortality of the triploid grass carp by 4 years (965 estimated fish in 1996). The project objective of 20-percent coverage of aquatic plants was not attained during the study period. Although the observed vegetation regrowth from 1 to 6 percent may have been accelerated by removal of 242 triploid

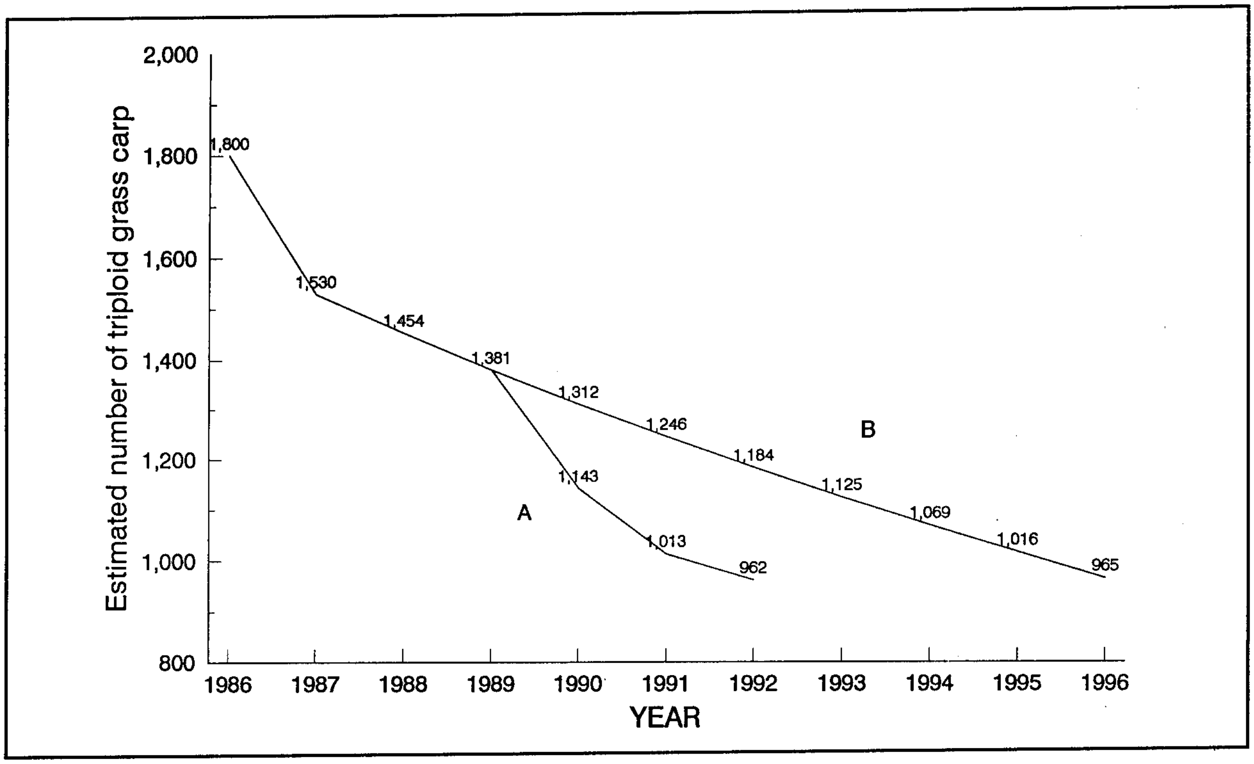

Figure 1. Estimated number of triploid grass carp in Lake Mills with angling (A) and projected without angling (B). Estimate is base on 15-percent mortality for the first year and 4 percent thereafter

grass carp, regrowth may have occurred even without the removal program.

Dedicated fishermen can remove a significant number of triploid grass carp, as demonstrated by three Lake Mills anglers who removed a total of 170 fish. Terrel and Fox (1974) reported angling removal of 429 grass carp in a Georgia lake (3.6 ha) over a period of 13 months. Although the fish were extremely crowded (253/hectare), this illustrates that up to 50 percent of the stocked population can be removed by angling.

Successful angling may be limited to lakes lacking abundant amounts of preferred food plants, such as Lake Mills. Results from an angling study in Washington (Bonar et al. 1993) demonstrated success in such lakes. However, grass carp were not attracted into baited areas nor were any caught in lakes containing abundant preferred food plants.

Some aspects of angling success on Lake Mills were comparable with those of smaller scale programs conducted on Lakes Waunatta and Winnemisett (Trent et al. 1992). On Lake Waunatta, a 27-ha private lake in Orange County, Florida, homeowners removed a total of 16 fish over a 4-month period. Catch rates were similarly higher initially, with 15 fish being caught during the first 3 weeks. As on Lake Mills, a small number of participants were responsible for the majority of the fish caught. Three individuals were responsible for a total of 13 fish. There were only three other successful anglers.

There were several notable points in the Lake Waunatta program. Although only 16 of the target of 50 fish were caught during the study period, the program was considered a success because plants preferred by triploid grass carp increased detectably during the study. A catch-per-unit effort of 0.5 fish per man-hour was achieved even though there were detectable levels of preferred food plants in the lake. Typical baits were successful, the exception being one fish caught on a strawberry.

On Lake Winnemisett, a 66-ha private lake in Volusia County, Florida, a total of 103 triploid

grass carp were removed, 70 by angling. A small number of participants, 4 out of a total of 10, were responsible for all 70 fish removed by angling. The remaining 33 fish were removed later by bowfishing and, incidentally, in a shiner lift net.

Public fishing is currently the most cost-effective triploid grass carp removal technique used by the GFC. This method may be useful in cases where a relatively small number of fish (less than 500) must be removed from a lake with low levels of preferred food plants, provided that there is sufficient angler participation. However, a more efficient and effective technique must be developed for removal of large numbers of fish, such as would be required in most lakes over 100 ha.

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Acoustic Fish Barriers for Grass Carp

by
Mike Curtin[1]

Sonalysts Corporate History

Sonalysts, Inc., is an employee-owned professional engineering services, research, and development organization. Sonalysts began in 1973 by offering extensive naval operational experience under contract to the U.S. Navy. Since that time, we have continually expanded and diversified our capabilities. Our extensive sonar system experience led to the development of our patented FishStartle System, which uses underwater sound to create barriers that fish avoid.

Sonalysts currently employs more than 400 people in offices throughout the United States. We have amassed an enviable record of completing over 6,000 contracts on time and within budget. For our commitment to excellence, we have been recognized by several agencies with many awards.

Successful Demonstrations

Sonalysts, Inc., is the first company to successfully deter fish with sound from both an operating nuclear power plant with 883 MW and 825 cfs of flow as well as a 30-MW hydroelectric project and 7,550 cfs of flow. Our system has also been used to keep fish away from a dredging and blasting site in Boston Harbor. In 1993-94, FishStartle was used on Staten Island at a thermal plant as well as at a hydroelectric plant on the Susquehanna River.

Nuclear power plant

In 1990, the James A. FitzPatrick (JAF) Nuclear Power Plant, located on Lake Ontario, was chosen by the Empire State Electric Energy Research Corporation (ESEERCO) to be the first operational site for a carefully conceived, scientifically performed, full-scale demonstration and feasibility study of an acoustic fish deterrence system. JAF had a documented impingement problem with alewives. The Nine Mile Point Nuclear Power Plant, also located on Lake Ontario, was chosen as the control facility. Nine Mile Point's intake is approximately 1 mile from the intake for JAF and provided an excellent comparison point for the purposes of the impingement study.

In 1991, Sonalysts patented FishStartle System was installed and operated at the JAF facility. During the test, independent contractors counted, not estimated, the number of alewives impinged at both facilities simultaneously. In addition, another independent contractor conducted hydroacoustic monitoring at the JAF intake to determine the density of alewives in front of the intake. Density measurements were based on established methods using state-of-the-art equipment. During the time when the FishStartle System was operated, JAF experienced an 87-percent reduction in alewife impingement as compared with the Nine Mile Point plant. A 96-percent reduction in fish density at the JAF intake was achieved as compared with the periods when the FishStartle System was not operating.

The success of the JAF demonstration clearly showed that Sonalysts FishStartle System is a very effective deterrent for alewives. The *North American Journal of Fisheries Management* article, dated Spring 1993, entitled "Response of Alewives to High Frequency Sound at a Power Plant Intake on Lake Ontario" documents this study.

[1] Sonalysts, Inc., Waterford, CT.

In 1993, our FishStartle System was installed and operated continuously from April 20 through July 19, which corresponds to the historical abundance period at JAF. The system achieved results similar to those reported in 1991. No evidence of acclimation was observed. A research paper, the third in a series describing this test, has been submitted for publication to the *North American Journal of Fisheries Management.*

Since the 1991 JAF experiment, Sonalysts has conducted and documented several other successful deterrence projects using FishStartle involving alewives, American shad, and blueback herring.

Hydroelectric generating station

In 1991 and 1992, Sonalysts conducted a demonstration of the FishStartle System at a hydroelectric generating station. The species of interest was the juvenile American shad. The initial purpose of the demonstration was to show that the FishStartle System would deter juvenile American shad from entering an area ensonified by the system. Underwater video cameras documented that the FishStartle System elicited the desired reactions from the shad. The FishStartle System was used to provide increased opportunities for shad to go downstream safely by keeping them away from turbine intakes and thus closer to the fish bypass pipe.

The demonstration made it clear that the overwhelming majority of American shad responded consistently and immediately every time the system was energized and sound transmitted into the water. The shad displayed no evidence of becoming acclimated to the sound source. Additionally, many more shad travelled down the fish bypass pipe when the system was transmitting than were noted at the fish bypass discharge when the system was off.

Marine construction and demolition project

In 1992, the annual spring migration of shad, blueback herring, and alewives coincided with underwater blasting of rock ledges in preparation to excavate a 3,850-ft trench for the immersed Third Harbor Tunnel between the city of Boston and Logan International Airport. Instead of halting marine work, Sonalysts deployed a mobile version of the FishStartle System and deterred fish from entering the blasting area, precluding a 3-month delay in the $5 billion project.

Current projects

Sonalysts is currently undertaking two large-scale demonstration projects with the FishStartle System. On Staten Island in New York, we will be demonstrating our unique capability at the Arthur Kill Generating Station. The species of interest are bay anchovy and blueback herring. This project began in August of 1993 and will continue through April of 1994.

In addition, Sonalysts deployed the FishStartle System at Metropolitan Edison's York Haven Station on the Susquehanna River in Pennsylvania. We used FishStartle to divert juvenile American shad away from the hydro plant's intake and toward a bypass gate. This project is now in the final report phase and is expected to conclude in early May 1994.

Sonalysts success

We believe the success Sonalysts has achieved in the application of acoustic fish deterrence is attributable to the following factors:

- Sonalysts, Inc., has more than two decades of experience in the field of underwater acoustics. Since 1973, Sonalysts has provided extensive technical, operational, and analytical support to the U.S. Navy for both submarine and surface ship sonar systems as a Department of Defense contractor. No other firm currently marketing acoustic fish deterrent systems or services can point to such extensive experience in the field of underwater sound.

- An integral part of our work with the Navy is developing underwater acoustic propagation models to determine not only how sound travels through the water column, but also what happens as it reflects from the surface and bottom. Using site-unique parameters, sound frequency, pressure levels, and target ensonification volumes, Sonalysts can use the models it has developed to design an optimum system for each site. Without this type of analysis, a costly trial and error approach can ensue, and the most effective system may never be implemented. No other firm currently marketing acoustic fish deterrent systems or services provides this measure of technical detail.

- Sonalysts, Inc., has assembled a team of experts including project managers, design engineers, field engineers, and physicists to successfully carry out this work.

Based on FishStartle's documented success with alewives, American shad, blueback herring, and coho salmon, Sonalysts is confident that we can successfully elicit an avoidance response from grass carp.

Publications

The success of Sonalysts FishStartle System has been documented in a number of professional journals and reports.

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Potential Applications for Grass Carp

Potential applications

There are many potential applications of FishStartle technology with grass carp. The most important application is probably to prevent grass carp from migrating into adjacent bodies of water. Other applications might be to keep grass carp in a specific area within a lake to reduce the required stocking density or to keep carp away from cooling water intakes. Regardless of the application, the initial phases of research are identical.

Confined area test

A confined area test is necessary to identify the optimum sound that grass carp avoid. This type of test is carried out in an area with deep, clear water. Fifty to one hundred carp

are placed within a cage and exposed to various underwater sound signals while underwater video is used to document their response. Once a sound signal is found that elicits the desired response, the test is repeated with various groups and ages of carp to ensure the results are repeatable.

Small-scale test

After identifying the optimum signal during the confined area test, a small-scale test is carried out to demonstrate that the system can be effective under operational conditions. A small site is selected to minimize costs. Therefore, the site selected for this test should be readily accessible for installing transducers and associated electronic equipment, allow for a method to easily quantify system effectiveness, and be acceptable to all interested parties. A variety of methods have been used for quantifying system effectiveness. For example, radio tags could be inserted in a number of fishes prior to releasing them near the barrier. A radio receiver is used to monitor the radio tags and determines if any fishes pass through the barrier. Alternatively, it is sometimes possible to deploy a net on the other side of the barrier to capture any carp that pass through the barrier. The method that is chosen is based on a trade-off of keeping costs down while still gathering useful data.

Full-scale demonstration

If the small-scale test is successful, a full-scale demonstration is often conducted on a larger site to for a longer term. The full-scale demonstration provides conclusive evidence that the system is effective for sites of all sizes under various environmental conditions.

Stocking Models and Potential Impacts

An Empirical Approach to Predicting Grass Carp Stocking Rates

by
Scott Bonar,[1] *G. L. Thomas*[1], *and Gilbert B. Pauley*[1]

Many factors can affect the level of aquatic plant control achieved using grass carp, including water temperature, target plant species, amount of plants present, grass carp mortality, and plant control objectives. Currently, most managers recommend only one or two grass carp stocking rates for lakes in their region based on the area coverage or biomass of vegetation and the management objectives for the site (Leslie et al. 1987; Seagrave 1988; Bonar 1990; Masser 1992). A single stocking rate may be adequate for specific geographic areas where little variation occurs in climate, target plant species, or management objectives; however, in mountainous regions where climate varies considerably or in areas where there are a wide variety of target plant species or management objectives, stocking rates must be adjusted to each site to be effective. To improve the ability to predict grass carp stocking rates for widely varying sites, we used regression analysis to examine the relationship between effective grass carp stocking rates and accumulated air temperature units for two major groups of plant species (preferred and less preferred) and for two types of management objectives (control of plants to a reduced level or eradication of all plants).

Two regression analyses were conducted based on the type of plant abundance data available. One (COVER) examined the relationship among stocking rate expressed in fish per vegetated hectare and the above-mentioned independent variables, while the other (BIOMASS) examined the relationship among stocking rate expressed in fish per metric ton of vegetation at the peak of the growing season and the same variables.

We obtained data for COVER from a questionnaire survey of grass carp managers in 28 states in the United States and 11 European countries (Bonar 1990). Data were not used from managers that had worked with grass carp for less than 3 years. For BIOMASS, we obtained stocking rates, associated data describing treatment sites and the results of the treatments from >100 grass carp field trials described in the literature.

Analysis of covariance revealed no significant differences in either area coverage-based or biomass-based stocking rates when rates were grouped by plant species preference (preferred or less preferred). Therefore, the data for both groups of plants were pooled to develop stocking rate curves based on the degree of plant removal: control and eradication.

The resulting four curves, COVER-control, COVER-eradication, BIOMASS-control, and BIOMASS-eradication, were evaluated for possible management applications. All relationships were best represented by curvilinear functions with high stocking rates necessary for cool regions declining to lower stocking rates for warmer areas. COVER-control (R^2 = 0.66), COVER-eradication (R^2 = 0.33), and BIOMASS-control (R^2 = 0.88) relationships were significant ($0.01<P<0.025$), while there was no discernible pattern in BIOMASS-eradication with temperature.

The curves can be used for prediction of stocking rates by selecting the curve with the appropriate management objective/plant abundance information and matching the accumulated air temperature units of the site to the

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corresponding location on the curve. The prediction obtained represents an average of successful stocking rates used by grass carp managers under similar conditions. Although curves were developed to predict stocking rates for either plant control or eradication, few sites (12 percent) examined in our study exhibited partial plant control 3 to 5 years following stocking, while the majority (88 percent) exhibited either eradication or no plant control. While the objectives of these treatments were often not stated, the low number of sites where partial control was achieved indicates that a specific level of aquatic plant control is currently difficult to obtain using grass carp.

We developed this approach based on worldwide data gathered using a variety of techniques. More precise predictions would be obtained by standardizing aquatic plant data collection, developing predictive curves for smaller geographic regions, and including other factors influencing the effectiveness of grass carp stocking rates, such as grass carp mortality, into the regression analysis.

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Simulation Model Evaluation of Sources of Variability in Grass Carp Stocking Requirements

by

R. Michael Stewart[1] *and William A. Boyd*[1]

Introduction

Grass carp have proven an effective control of nuisance aquatic plant growth since their introduction into the United States in 1963 (Guillory and Gassaway 1978). Grass carp were initially banned from many States because of the potential ecological risk associated with releasing reproductively viable diploid variants (Stanley, Miley, and Sutton 1978). Though the advent of reliable triploid inducement techniques (Cassani and Caton 1986) has lessened the ecological threat stemming from releases of reproductively viable diploids, grass carp use is still restricted in large open waterways by some States and prohibited altogether by others (Allen and Wattendorf 1987).

Part of the reason for continued restrictions on grass carp use is concerned with the uncertainty in determining stocking strategies that provide desired control levels without risking unwanted impacts (Noble, Bertolli, and Bestill 1986; Leslie et al. 1987). Table 1 illustrates the extreme variability in stocking rates that have been used under different conditions.

Table 1
Stocking Rate Requirements for Case Studies

Source	State	Stocking Rate, fish/vegetated ha
Sutton and Van Diver (1986)	Florida	3 to 638
Leslie et al. (1987)	Florida	9 to 440
TVA (1990)	Alabama	17
Klussman et al. (1988)	Texas	74
Santha et al. (1991)	Texas	79 to 130
Wiley et al. (1984)	Illinois	47 to 370
Bonar et al. (1993)	Oregon	180
Stocker and Hagstrom (1985)	California	6,323

Factors accounting for variability in reported stocking rates include control objectives, water body characteristics (e.g., size, temperature regime, and geographic region), plant infestation characteristics (e.g., plant species, overwintering level, regrowth rate, and peak density), and grass carp stocking size, mortality, and feeding and dispersal behavior.

Because of the multiple issues involved, determination of proper stocking rates must be based on individual site conditions and objectives. Further, because grass carp are long-lived and difficult to remove after stocking, overstocking can create long-term undesirable impacts (Leslie et al. 1987; Klussman et al. 1988). In efforts to aid determination of proper stocking rates, several computer models have been developed by various State and Federal agencies in the past (Miller and Decell 1984; Swanson and Bergersen 1988; Wiley et al. 1984; Boyd and Stewart 1990; Santha et al. 1991). In general, each of these models represents a simplified account of the overall processes that interact within this complex biocontrol system. They by no means have been designed to consider all of the biological or environmental variability encountered under real conditions. Instead, these models provide a desktop tool for examining the effects of user defined variability in one or more of the identified system drivers. Through exercise of these tools, aquatic plant control decision makers are provided the opportunity to ask "What if" type questions regarding grass carp use under environmental and biotic conditions representative of their water body.

The objectives of this paper are to illustrate use of the White Amur Stocking Rate Model

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developed by the U.S. Army Engineer Waterways Experiment Station (WES) for evaluating effects of selected site conditions on stocking rate requirements.

Model Description

The current version of the WES White Amur Stocking Rate Model (AMUR/STOCK) is based in part on an earlier model developed by WES (Miller and Decell 1984) and on the grass carp bioenergetics module included in the Illinois Herbivorous Fish Simulation System (Wiley et. al 1985). Along the lines of these two earlier simulations, the current WES model has separate routines for generating daily estimates of plant biomass and for estimating the size and remaining number of grass carp.

The plant growth simulation routine considers the effects of plant species, season, temperature, current plant biomass, and previous herbivory level on daily plant growth and mortality rates. Daily net growth is calculated by subtracting mortality losses (leaf sloughing) from the daily growth increment. Daily net growth, which is also corrected for a fish herbivory estimate generated by the fish routine, can therefore be negative in value. A daily update of total plant biomass is calculated by adding the daily growth increment to the total plant biomass estimate for the previous day.

Daily plant consumption by grass carp is determined by the number of grass carp remaining and relationships that consider the average size of fish, temperature, and plant species. Daily fish growth is calculated from the daily consumption amount, which is adjusted to account for assimilation efficiency of consumed plant material. Assimilation efficiency is also a function of fish size, temperature, and plant species. Finally, fish metabolic costs are subtracted from the assimilated amount. All calculations in the fish bioenergetics module are based on caloric values, which are estimated for each plant species. Final conversion of net caloric gain into fish biomass is a function of previous fish size and temperature. Reductions to the number of fish are made at the end of each year of the simulation period. Yearly mortality estimates can be input by the user, or the model can be run using a default mortality setting of 10 percent per year.

Model Application Procedures

The first step in applying the model is calibration of the plant growth module for the plant infestation in question. This is a two-step process that involves initializing the module for the plant species, the overwintering plant biomass level, and the peak plant biomass level. These biomass estimates should be provided from field measurements. Next, the user must make adjustments to the calibration datasets for the effects of season and temperature on daily plant growth and mortality rates. These calibration datasets must be corrected in order that calculated plant growth rates generate a seasonal plant growth curve that represents field conditions. This process should include input of a water temperature dataset for the water body to be evaluated. If temperature datasets are not available to the user, the model can be initialized with one of several default datasets included with the model software package.

Initialization of the grass carp module includes inputs for the number of grass carp to be stocked and the average size of individual fish. Finally, the user inputs the duration of the simulation period and a fish mortality estimate for each year of the simulation period.

After first generating an estimate of the number of hectares of control for each year of the simulation period that the stocked fish will exert on the target plant infestation, the model next calculates the initial stocking rate requirement (fish per vegetated hectare) for obtaining control during each subsequent year. For example, consider a simulation under which 1,000 grass carp were estimated to control 10 ha of vegetation at the end of Year 1, 50 ha by Year 2, 200 ha by Year 3, 500 ha by Year 4, and 1,000 ha by Year 5. Final model calculations would estimate that initial stocking rates of 100 fish per vegetated

acre would provide control by the end of Year 1. If control objectives were to obtain control within the first year, a stocking rate near this level would be considered appropriate for the simulation conditions. However, if control objectives provided that control was not necessary until the third year after stocking, then initial stocking rates could be reduced to five fish per vegetated hectare (i.e., 1,000 fish stocked at 200 ha controlled by Year 3).

Demonstration of Model Use

General initializations

As stated above, the objectives of this paper are to demonstrate use of the WES White Amur Stocking Rate Model for evaluating effects of selected site conditions on stocking rate requirements. For these comparisons, certain initialization conditions were constant for all simulation runs. The water temperature dataset illustrated in Figure 1 was taken from average median monthly water temperatures from Guntersville Reservoir, Alabama, for the 4-year period, 1984-1987. Note shading in Figure 1 that highlights those times of the year when water temperatures are below 12 °C, the lower threshold temperature for grass carp feeding. Average size of stocked fish was initialized at 0.34 kg, an approximation of the average weight of a 30-cm fish (Kirk, Morrow, and Killgore 1994). All simulations additionally considered that the target plant was *Hydrilla verticillata*, a plant that ranks highly on most feeding preference lists of grass carp (Sutton and Van Diver 1986; Leslie et al. 1987). Finally, grass carp mortality rates were set to zero for simulations generated to evaluate the effects of different levels of peak and overwintering plant biomass.

Levels of peak biomass

Peak biomass levels evaluated for their effects on grass carp stocking requirements were 110, 220, and 440 g/m^2. These values equate on a fresh weight basis to approximately 11, 22, and 45 metric tons per hectare, respectively. In field situations, these differences in peak biomass levels could occur because of different plant species or for a given

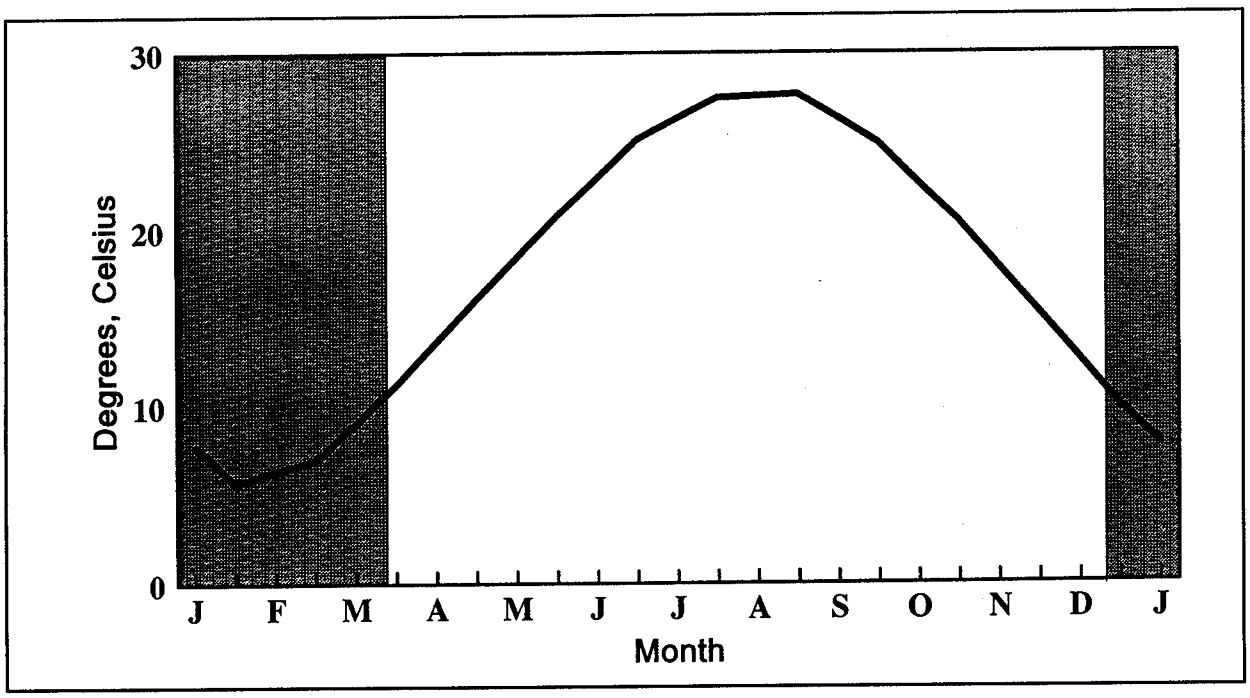

Figure 1. Water temperature file derived from average median monthly water temperatures for Nickajack Dam tailrace, Guntersville Reservoir, Alabama, during 1984-1987. Shading indicates periods when temperatures were below 12 °C, the lower threshold temperature for grass carp feeding in the model

plant species because of differences in colony age, prior treatment history, or for a list of environmental factors, including water depth, light availability, and sediment nutrient levels.

For each of these three peak biomass levels, plant growth was initiated with an overwintering biomass level of 11 g/m^2, or approximately 1 metric ton per hectare. The resulting plant growth curves, generated using different calibration datasets for the effects of season and temperature on growth rates, are shown in Figure 2. Note that the spring regrowth rate for the high peak biomass growth curve is the highest, as illustrated by this growth curve having a greater slope during the spring period.

Levels of overwintering biomass

Overwintering biomass levels can also vary among plant species, geographic region, water body type, and location within a water body. Overwintering biomass levels considered herein were 11, 44, and 110 g/m^2. The lower level represents plant colonies that overwinter as root crowns, specialized overwintering structures (e.g., tubers) or nonintact stem fragments. In these plant colonies, the majority of shoot material generated during the growing season is lost through senescence or by other processes (e.g., displacement because of hydrological breakage and transport). The highest overwintering level represents plant colonies that overwinter intact. As stated above, these differences in overwintering levels can be due to differences in either plant species or site conditions.

Peak plant biomass for the simulations representing the three overwintering biomass levels was initialized at 440 g/m^2. Resulting plant growth curves are illustrated in Figure 3. In comparison, note that although regrowth rates were similar for the three simulations, attainment of peak biomass was delayed by approximately 2 months in the simulation for the lowest overwintering biomass level.

Levels of grass carp mortality

Differential mortality levels in stocked grass carp are a major source of variability in stocking effectiveness. Numerous factors account for mortality variability, including differences

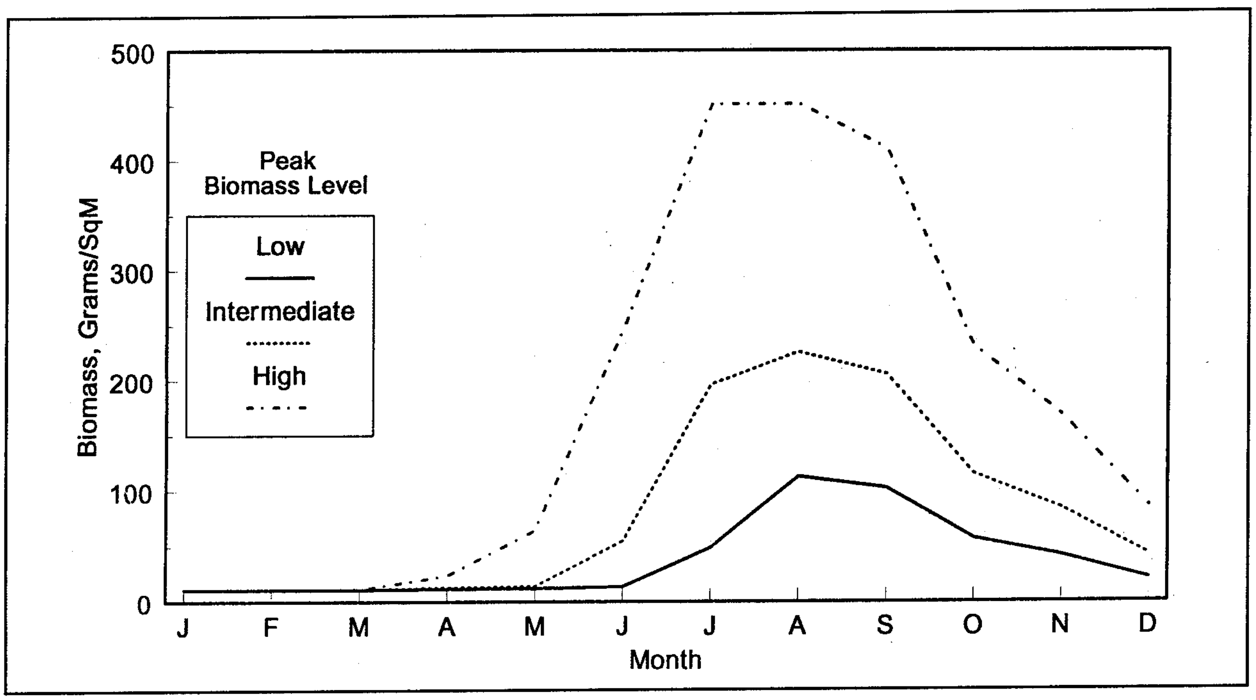

Figure 2. Seasonal plant biomass curves generated by the AMUR/STOCK model for considering the effects of three different peak biomass levels on grass carp stocking rate requirements. Refer to text for initialization conditions

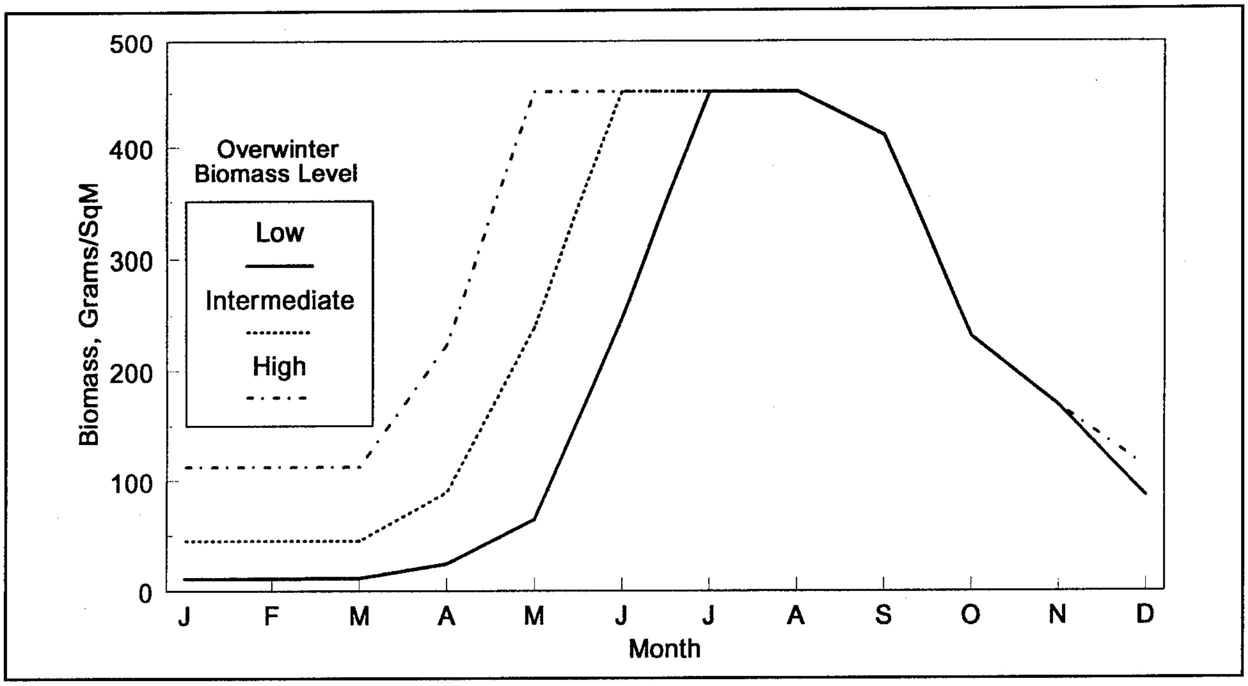

Figure 3. Seasonal plant biomass curves generated by the AMUR/STOCK model for considering the effects of three different overwintering biomass levels on grass carp stocking rate requirements. Refer to text for initialization conditions

in stocking size, predator populations, and commercial fishing pressure. In addition, movement of stocked fish results in variable losses from the target treatment area over time. To illustrate the significance of fish losses, simulations for a 10-year period representing three levels of grass carp mortality were generated. As a baseline for comparison, one set of simulations considered a zero-mortality level for each year of the 10-year period. The second set of simulations used the model default setting of 10-percent mortality per year. The highest mortality settings for these comparisons considered 50-percent mortality in Year 1, 25 percent in Years 2 and 3, and 10 percent in Years 4 through 10.

Simulation Outputs

For the three sets of conditions described above, simulation outputs were generated to provide estimates of the stocking rates (April of Year 1) required to provide control by late summer of poststocking Years 1 through 10. Stocking rate requirement estimates generated for the effects of peak plant biomass are provided in Table 2. As was expected, stocking rate requirements were sensitive to peak biomass level as well as to poststocking time. Stocking requirements ranged from 167 fish/ha for control of high peak biomass plant growth by Year 1 to 4 fish/ha for control of low peak plant biomass growth by Years 6 through 10. Stocking requirements for the high peak biomass plant growth simulations were consistently highest for each poststocking year. For control in the first poststocking year, rates

Table 2
Stocking Rate Requirements for Different Peak Biomass Levels

Poststocking Year	High	Peak Biomass Level Intermediate	Low
1	167	71	32
2	31	22	12
3	16	13	7
4	12	9	6
5	10	8	5
6	9	7	4
7	9	7	4
8	9	7	4
9	8	7	4
10	8	7	4

estimated for high peak biomass plant growth were five times higher than for low peak biomass plant growth. This difference had decreased to a two-fold higher rate by the fifth poststocking year. Differences in stocking rates remained approximately two-fold for the remainder of the 10-year period.

Stocking rate requirements for the three overwintering plant biomass levels are given in Table 3. Increases in overwintering biomass resulted in significant increases in stocking rate requirements. Stocking requirements in the simulations ranged from 526 fish/ha to 8 fish/ha, depending on overwintering biomass levels and poststocking year. The stocking rate requirement for the high overwintering biomass level was approximately three times higher than the low level in each poststocking year. These simulation outputs are of further significance when compared with simulation outputs for low peak biomass described above. Under these comparisons, stocking requirements for Years 7 through 10 for the high overwintering biomass plants, which were initialized with the highest peak biomass level, are approximately 6 to 7 times higher than requirements for low peak biomass plants, which were initialized with the lowest overwintering biomass level.

Table 3
Stocking Rate Requirements for Different Overwintering Biomass Levels

Poststocking Year	High	Overwintering Biomass Level Intermediate	Low
1	526	222	167
2	95	42	31
3	50	22	16
4	37	16	12
5	31	14	10
6	29	13	9
7	27	12	9
8	26	12	9
9	26	11	8
10	25	11	8

Simulation outputs of stocking rate requirements for the three grass carp mortality levels are given in Table 4. Outputs for "zero mortality" simulations indicate that stocking rate requirements will decrease annually over the first 4 years and then stabilize for the remainder of the 10-year simulation period. Increases in grass carp mortality rates resulted in increased stocking rate requirements for each poststocking year. For example, in order to maintain control into the tenth poststocking year, Year 1 stocking rates would have needed to have been approximately seven times higher under the high mortality conditions (58 fish/ha) than under the zero-mortality conditions (8 fish/ha). It is further noteworthy that peak effectiveness (i.e., minimum stocking requirements) for simulations that experienced grass carp mortality losses occurred in poststocking Years 5 to 6, after which time stocking rate requirements gradually increased.

Table 4
Stocking Rate Requirements for Different Grass Carp Mortality Levels

Poststocking Year	Zero	Grass Carp Mortality Level Intermediate	High
1	167	169	175
2	31	36	68
3	16	21	48
4	12	17	45
5	10	16	42
6	9	16	43
7	9	17	45
8	9	19	48
9	8	20	53
10	8	22	58

Management Considerations

Numerous interacting factors account for variability associated with grass carp effectiveness following release. The effects of these factors therefore must be considered in order to determine proper stocking rates for the desired level of aquatic plant control. The WES Stocking Rate Model was designed to help aquatic plant managers consider the effects of some of these factors. Examples presented herein illustrate the significance of poststocking time coupled with two characteristics of plant growth (i.e., overwintering and peak biomass levels) and with grass carp mortality rates.

Poststocking time was the single most important factor affecting stocking rate requirements in these simulations. For the different

sets of simulation conditions considered, stocking rate requirements were as much as 20 times higher for control by the end of the first year than by the end of the sixth and subsequent years. These differences stem from the fact that as fish grow in size, they eat more, and therefore fewer are required to provide the same level of control. Leslie et al. (1987) advised caution in selecting stocking rates that would provide control of hydrilla in the first couple of years in multiuse lakes. As peak effectiveness of stocked fish is often delayed until the fifth or sixth year following stocking, stocking rates aimed at providing control of all target areas within the first few years following stocking will probably be too high. Results of such improper stocking would include both unnecessary costs for purchase of fish and possible detrimental ecological impacts to nontarget areas.

Generally speaking, grass carp are effective only when they can consume more plant biomass than is produced by plant growth in the target area. Because plant growth varies among different plant species and site conditions, researchers often recommend that stocking rates be based on numbers of fish per metric ton of vegetation (Leslie et al. 1987). Though this method is more advantageous than stocking rates based solely on area of infestation, simulation outputs herein indicate that overwintering biomass levels and regrowth rates should also be considered in addition to peak biomass levels.

In addition to plant growth characteristics, grass carp losses also affect stocking results. Variability in loss rates are known to occur because of differences in grass carp health condition or size at stocking or to differences in predator populations. Additionally, similar losses could result from nonmortality-based factors, such as off-target movement within the stocked water body or escape from the water body. No matter what the cause in grass carp losses, the net result is decreases in plant consumption and control effectiveness. In situations where actual losses are high and cannot be reduced by stocking technique, higher stocking rates will be required.

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Aquatic Vegetation and Water Quality in Lake Marion, South Carolina

by
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Introduction

Dense stands of aquatic vegetation have a significant effect on water quality (Buscemi 1958; Ultsch 1973; Schreiner 1980; Wylie and Jones 1987) and fish habitat (Engel 1988). Dense growth of aquatic vegetation in a reservoir can decrease the amount of vertical mixing that occurs in the water column, decrease lateral flow and mixing, reduce turbulence and aeration, and promote thermal stratification. Buscemi (1958) found dissolved oxygen (DO) to be lower and more sharply stratified under dense beds of *Egeria densa* than in adjacent open water. Water temperatures were higher and DO concentrations lower under extensive surface mats of water hyacinth (*Eichhornia crassipes*) than in open-water areas of experimental ponds (Ultsch 1973; Schreiner 1980). Wylie and Jones (1987) related diel and seasonal changes of DO and pH to the aquatic macrophyte community of a shallow reservoir in southeast Missouri. It is generally recognized that aquatic vegetation, when present in significant amounts, has an important influence on the chemical and physical environments of impoundments.

Lake Marion, South Carolina, is a 44,000-ha impoundment composed of open water, standing submerged trees, and thick cypress swamps. Aquatic vegetation has become a serious problem in upper Lake Marion north of the Interstate-95 (I-95) bridge (Inabinette 1985). The shallow, nutrient-rich upper Lake Marion area is conducive to aquatic plant growth. Many areas of the lake have limited access because of extremely thick vegetation. This has hampered use of the lake by recreational hunters, anglers, and boaters. Lake Marion has a long history of nuisance aquatic vegetation problems, dating back to large-scale alligator weed (*Alternanthera philoxeroides*) infestation in the 1940s (Inabinette 1985). Water quality studies have been completed on both Lake Marion and the Santee Swamp (Inabinette 1985; Harvey, Pickett, and Bates 1987; Bates and Marcus 1989). There have also been assessments of the distribution of aquatic vegetation on Lake Marion (Welch, Fung, and Remillard 1985) and the species of vegetation present in the lake (Inabinette 1985). No studies have attempted to determine the relationship between aquatic vegetation and water quality in Lake Marion, nor have the seasonal changes in species composition of aquatic vegetation been investigated.

Three hundred thousand triploid grass carp (*Ctenopharyngodon idella*) were stocked in upper Lake Marion between 1989 and 1991 for the purpose of vegetation control (South Carolina Aquatic Plant Management Council and South Carolina Water Resources Commission 1989). Current data are needed in order to assess the impact of this measure on water quality and aquatic vegetation. The objectives of this study were to (a) determine the changes in aquatic vegetation abundance and water quality over the course of 1 year at five different areas in upper Lake Marion, (b) determine the seasonal changes in aquatic vegetation species composition at these five areas, and (c) determine the relationship between water quality and aquatic vegetation abundance in these five areas.

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Study Area

Lake Marion was formed in 1941 by the impoundment of the Santee River (Inabinette 1985). Wateree and Congaree rivers join to form Santee River about 15 km upstream from Lake Marion. Lake Marion is a eutrophic reservoir with an average depth of 4 m and a maximum depth of 23 m (Inabinette 1985).

Upper Lake Marion, the defined study area, is north of the I-95 bridge. This area has standing submerged trees, open shallow flats, and thick stands of bald cypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*). Upper Lake Marion has an estimated 4,800 ha of submerged vegetation. Santee Swamp is immediately upstream from upper Lake Marion and covers approximately 6,500 ha. The swamp is anaerobic for most of the year and affects the water quality of upper Lake Marion (Bates and Marcus 1989).

Five sampling areas were selected as representative of different habitat types present in the upper Lake Marion area (Figure 1). Three of the areas, Pack's Flats, Elliot's Flats, and

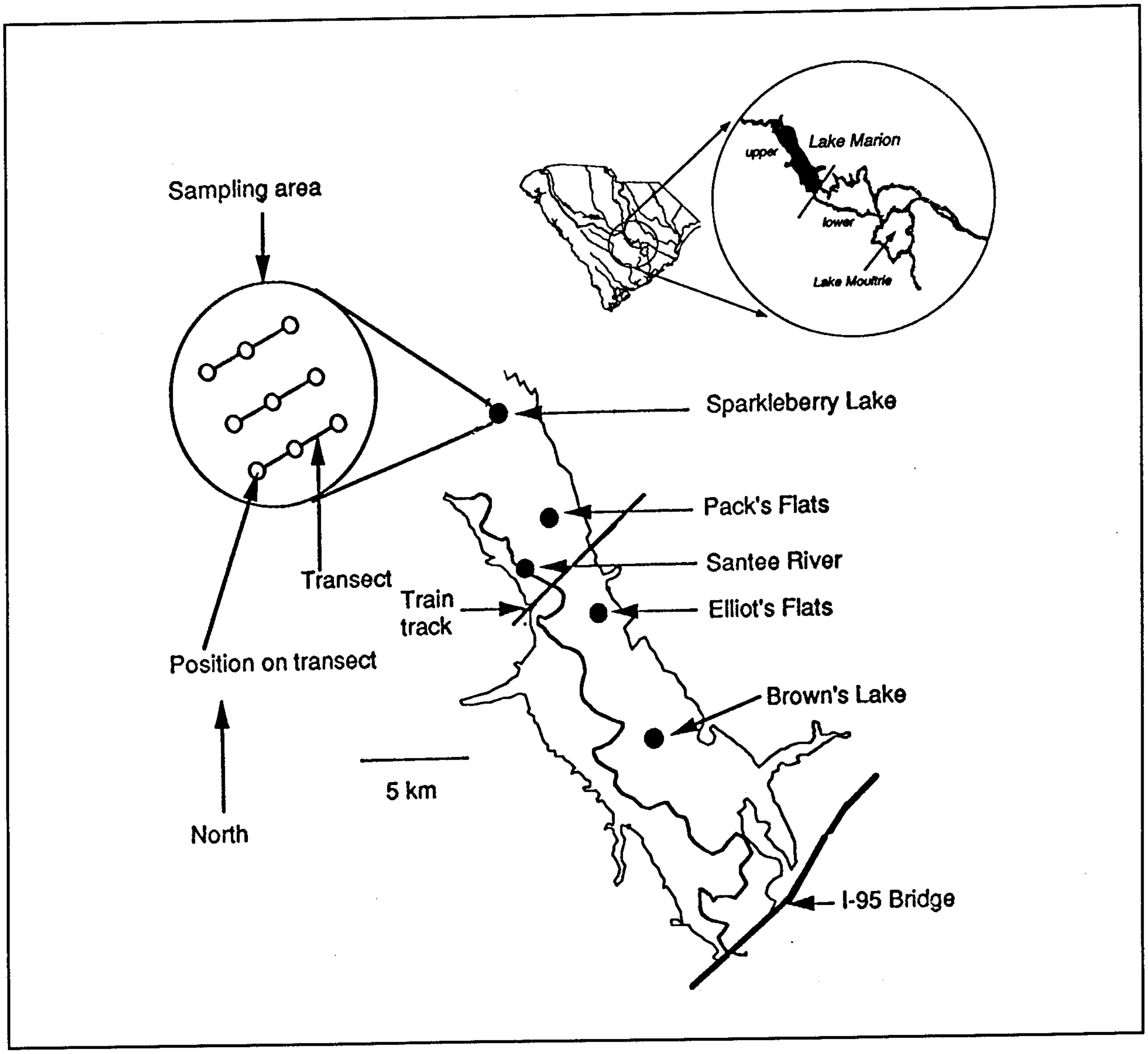

Figure 1. Location of upper Lake Marion on a map of South Carolina (top). Location of study areas in upper Lake Marion and a diagram of transects and positions in an area

Brown's Lake, were in upper Lake Marion proper. Pack's Flats is a shallow (depth = 1.0 to 2.0 m) open area in the uppermost region of Lake Marion near the Santee Swamp; hence water quality in Pack's Flats is strongly influenced by the Santee Swamp (Bates and Marcus 1989). Elliot's Flats sampling area is located approximately 3.5 km southeast of Pack's Flats. This area is slightly deeper (depth = 1.5 to 2.5 m) than Pack's Flats and receives water directly from the Santee River via a canal as well as some flow from smaller tributaries and the Santee Swamp. Brown's Lake is located approximately 7.5 km southeast of Elliot's Flats and approximately 1 km below the confluence of Lake Marion and the Santee River channel. It is a relatively deep area (depth 3.5 to 4.5 m) characterized by standing submerged trees. The other two sampling areas, Sparkleberry Lake and Santee River, are located in the Santee Swamp and in the Santee River, respectively. Sparkleberry Lake is a small (1 ha), shallow (depth = 1.0 to 1.5 m) lake in the lower end of the Santee Swamp. Santee River sampling area is located in the Santee River, channel 9 km above its confluence with Lake Marion. All of these areas experience considerable seasonal changes in water level and flow.

Methods

The five study areas were sampled biweekly for 12 months from January to December 1991. Three parallel transects 60 m long were marked 100 m apart in each area. Aquatic vegetation abundance and primary and secondary vegetation species were recorded at three equidistant positions along each transect for a total of nine readings per area per biweekly sampling period (Figure 1). DO concentrations, water temperature, and conductivity were also recorded at the surface and bottom at each position. Vegetation abundance was recorded as one of five categories for each position. Each category was assigned a numerical value. The categories and their numerical values were as follows: 0—no vegetation present, 1—vegetation sparse, 2—vegetation present but submersed, 3—vegetation surface coverage less than 50 percent, 4—vegetation surface coverage greater than 50 percent. Abundance values were summed for each transect resulting in three abundance values per area per sampling period, each with values from 0 to 12. Differences in vegetation abundance among biweekly sampling periods and differences in abundance among study areas were tested in separate one-way analysis of variance tests (ANOVA) (Harvey 1982; SAS Institute, Inc. 1985). A t-test on least squares means (Harvey 1982) was also performed to determine which areas differed significantly from other areas.

An aquatic vegetation sample was taken at each position with a rake. Primary vegetation species was the species that comprised the largest proportion of the sample. Secondary vegetation was the species comprising the next largest proportion. Vegetation was identified in the field (Aulbach-Smith and DeKoslowski 1990). Sampling periods were combined into four seasons to examine seasonal differences in primary vegetation species composition. The seasons were defined as follows: winter (December 21 - March 20), spring (March 21 - June 20), summer (June 21 - September 20), fall (September 21 - December 20). Two categories of primary vegetation were assigned numerical values: 1—*Hydrilla* and 0—other vegetation species. These two categories were used for the purpose of statistical analysis because of the predominance of *Hydrilla* as the primary vegetation species. The mean of the values was calculated for each area during each sampling period. This mean, which was between 0 and 1, represented the proportion of each area in which *Hydrilla* was the primary species of vegetation. Differences in this proportion among seasons and among areas were then analyzed using ANOVA (SAS Institute, Inc. 1985) and a t-test on least squares means (Harvey 1982). Secondary vegetation species were recorded at each position but were not included in statistical analysis. The Santee River study area was omitted from statistical analysis procedures on vegetation species composition and abundance because of the year-round absence of aquatic vegetation in the river channel.

DO concentration (to nearest 0.1 mg/L), water temperature (to nearest 0.1 °C), and conductivity (to nearest µmho/cm) were measured 10 cm below the surface and 10 cm above the bottom at each position. DO and temperature were measured with a Yellow Springs Instruments DO meter (Model 51 B).

Differences in DO concentrations, water temperature, and conductivity among areas and among sampling periods were analyzed using ANOVA and a t-test on least squares means. Differences between surface and bottom DO concentrations, surface and bottom temperature, and surface and bottom conductivity values were calculated for each area and a t-test performed with the null hypothesis that the difference = 0.

Pearson correlation coefficients were calculated for vegetation abundance and surface and bottom DO concentrations, temperatures, and conductivity values to determine if these parameters were correlated. All error testing was done at alpha = 0.05 level.

Water temperature and DO readings were taken at 30-cm intervals from surface to bottom on 17 September (week 36) and during July of 1992 and 1993 to examine differences in water quality throughout the water column in summer. Readings were taken at three positions in each area and the average of the three readings is reported. These readings were taken at all study areas except the Santee River.

Results

Vegetation abundance differed significantly among all study areas ($F = 184.3$, $p \leq 0.0001$) except Brown's Lake and Sparkleberry Lake ($p = 0.77$). Relative abundance rankings (on a scale of 0 to 12) averaged for the year were as follows: Pack's Flats = 9.12, Elliot's Flats = 7.13, Sparkleberry Lake = 2.43, Brown's Lake = 2.27. Relative abundance rankings differed significantly among sampling periods at Elliot's Flats ($F = 7.45$, $p \leq 0.0001$) and Pack's Flats ($F = 18.06$, $p \leq 0.0001$), but not at Brown's Lake ($F = 0.96$, $p = 0.5212$) or Sparkleberry Lake ($F = 1.61$, $p = 0.0822$). Figure 2 illustrates

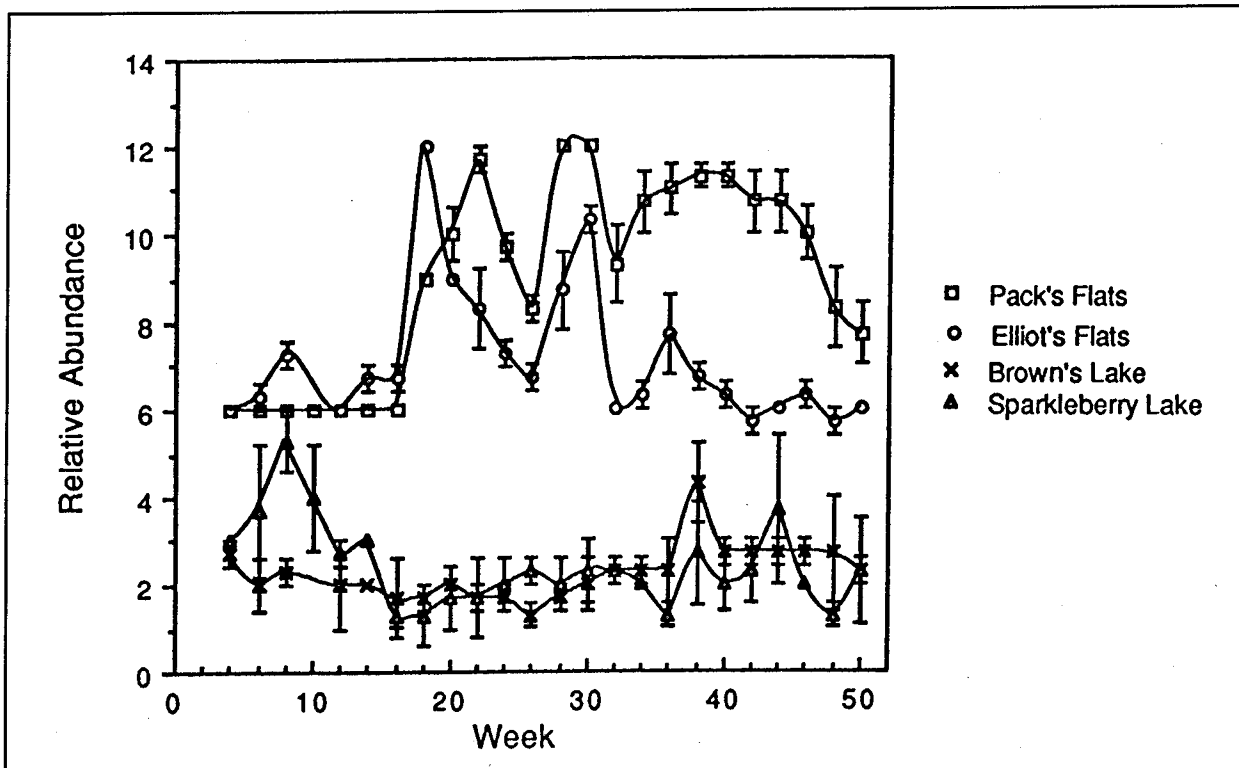

Figure 2. Relative vegetation abundance at four study areas in upper Lake Marion,South Carolina, during 1991

the changes in vegetation abundance over the course of the study. Brown's Lake had very little aquatic vegetation throughout the year, and a surface canopy of vegetation never formed. Sparkleberry Lake had small patches of both submersed and emergent vegetation throughout the year, but was never heavily infested. Sparkleberry Lake had slightly higher abundance values in late fall, winter, and early spring when there was little shading from the canopy of cypress and water tupelo trees. In contrast, Pack's Flats and Elliot's Flats were heavily infested with *Hydrilla* and other species of aquatic vegetation throughout the year. In the winter, aquatic vegetation was mostly submersed. As water temperatures and the photoperiod duration increased in spring and summer, vegetation increased, and a dense surface canopy formed at both areas. Elliot's Flats and portions of Pack's Flats were treated with aquatic herbicide by Santee Cooper Public Service Authority (to facilitate recreational access) during week 18 and week 29 of the study. Vegetation abundance in these two areas declined in late fall and winter as water temperature and the photoperiod duration decreased.

Vegetation abundance was significantly correlated with the following: surface DO ($R = 0.1977$, $p \leq 0.0001$), bottom DO ($R = -0.2717$, $p \leq 0.0001$), surface temperature ($R = 0.2172$, $p \leq 0.0001$), bottom temperature ($R = 0.1205$, $p = 0.0006$), surface conductivity ($R = 0.2392$, $p \leq 0.0001$), and bottom conductivity ($R = 0.2294$, $p \leq 0.0001$).

Hydrilla was the primary vegetation for 66 percent of the observations taken during the study (Figure 3). Other vegetation species included water primrose (*Ludwigia uruguayensis*), *Egeria densa*, coontail (*Ceratophyllum demersum*), *Potamogeton* spp., *Najas* spp., and *Bidens* spp. Primary vegetation differed significantly among all study areas ($F = 795.62$, $p \leq 0.0001$) except Pack's Flats and Elliot's Flats ($p = 0.13$). Primary vegetation did not differ significantly among seasons ($F = 1.58$,

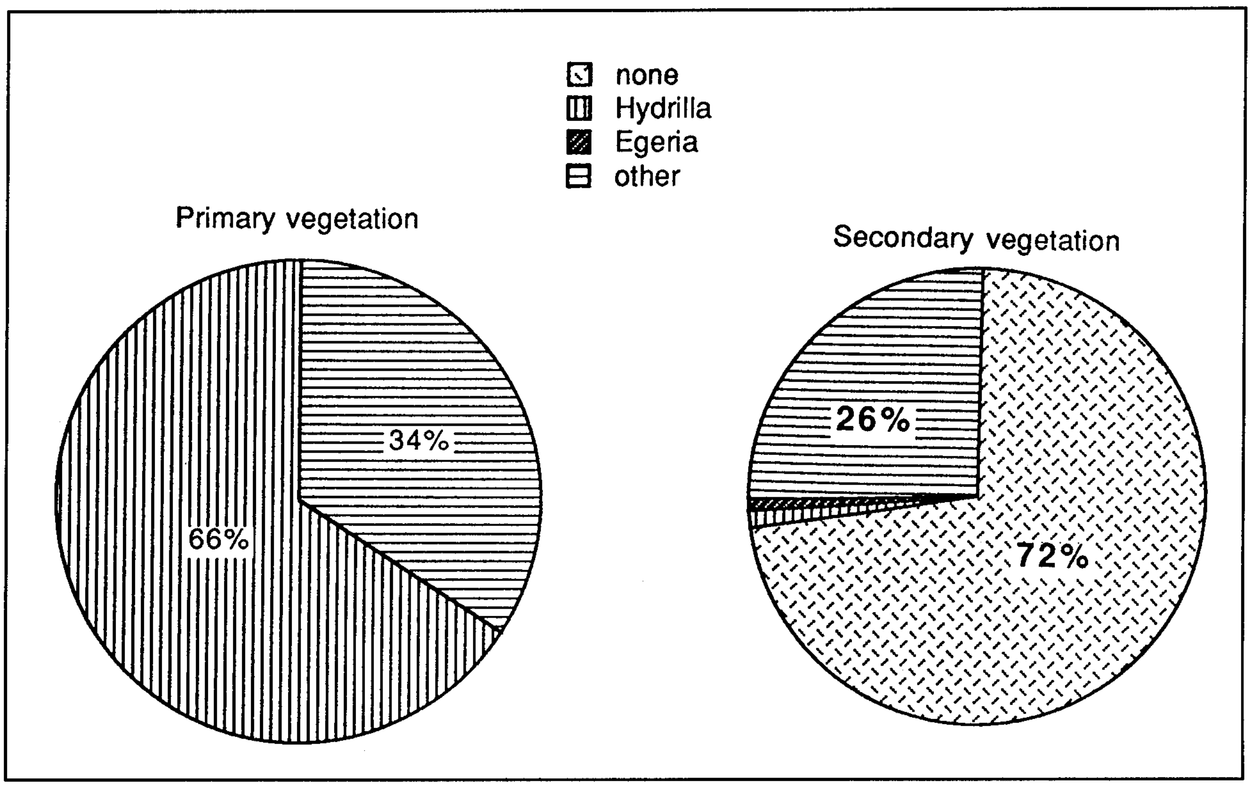

Figure 3. Primary and secondary species of aquatic vegetation at four study areas in upper Lake Marion, South Carolina, during 1991

p = 0.2003). No secondary vegetation was present for 72 percent of the observations (Figure 3). *Hydrilla* was the secondary vegetation species for only 1 percent of the observations. Other species of secondary vegetation are listed above.

DO levels were highest during January (week 2) and generally decreased throughout the year until late September (week 38) (Figures 4-8). At this time, DO concentrations began to rise and continued to do so through December (week 48). Lowest DO concentrations recorded for all areas were during the months of July, August, and early September. Surface DO levels differed significantly from bottom DO levels for all study areas ($t_{\text{Brown's Lake}} = 3.34$, $p \leq 0.0001$; $t_{\text{Elliot's Flats}} = 8.49$, $p \leq 0.0001$; $t_{\text{Pack's Flats}} = 13.00$, $p \leq 0.0001$; $t_{\text{Sparkleberry Lake}} = 9.19$, $p \leq 0.0001$; $t_{\text{Santee River}} = 8.68$, $p \leq 0.0001$).

Surface DO levels differed significantly among all study areas ($F = 80.40$, $p \leq 0.0001$) except Pack's Flats (Figure 4) and Santee River (Figure 5) ($p = 0.11$). Surface DO levels also differed significantly among sampling periods ($F = 99.51$, $p \leq 0.0001$). Elliot's Flats (Figure 6) and Pack's Flats areas exhibited the greatest biweekly fluctuations. Highest mean surface DO concentration recorded was 11.2 mg/L at Pack's Flats on 19 February (week 6) and 28 May (week 20) and at Sparkleberry Lake (Figure 7) on 19 March (week 10). Lowest mean surface DO concentrations recorded were 0.7 and 0.8 mg/L at Sparkleberry Lake on 11 July (week 26) and 03 September (week 34), respectively. Surface DO concentration was related to surface temperature and vegetation abundance (R-square = 0.91) (Figure 9). Surface DO concentrations increased as vegetation abundance increased and were highest at or below 10 °C and above 20 °C.

Bottom DO concentrations were usually lower than those at the surface and did not exhibit as much biweekly fluctuation (Figures 4-8). Bottom DO concentrations differed significantly among all study areas (F = 240.88,

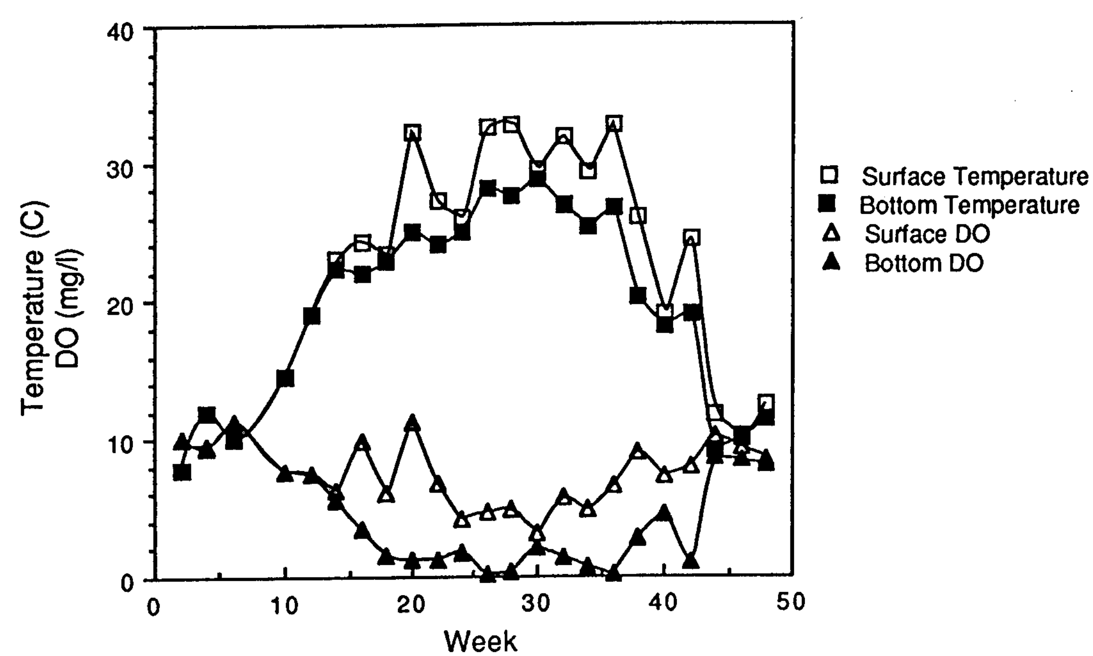

Figure 4. Surface and bottom temperature and DO levels at Pack's Flats, Lake Marion, South Carolina, during 1991

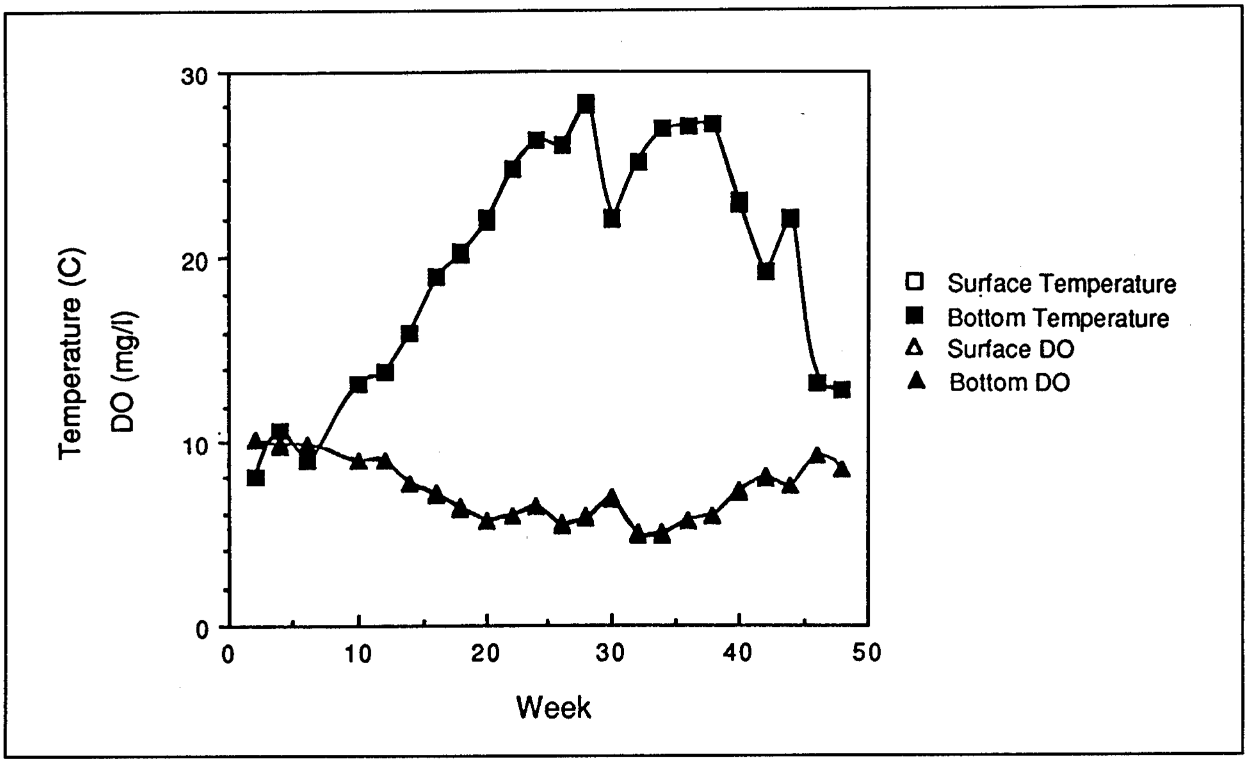

Figure 5. Surface and bottom temperature and DO levels in Santee River, South Carolina, during 1991

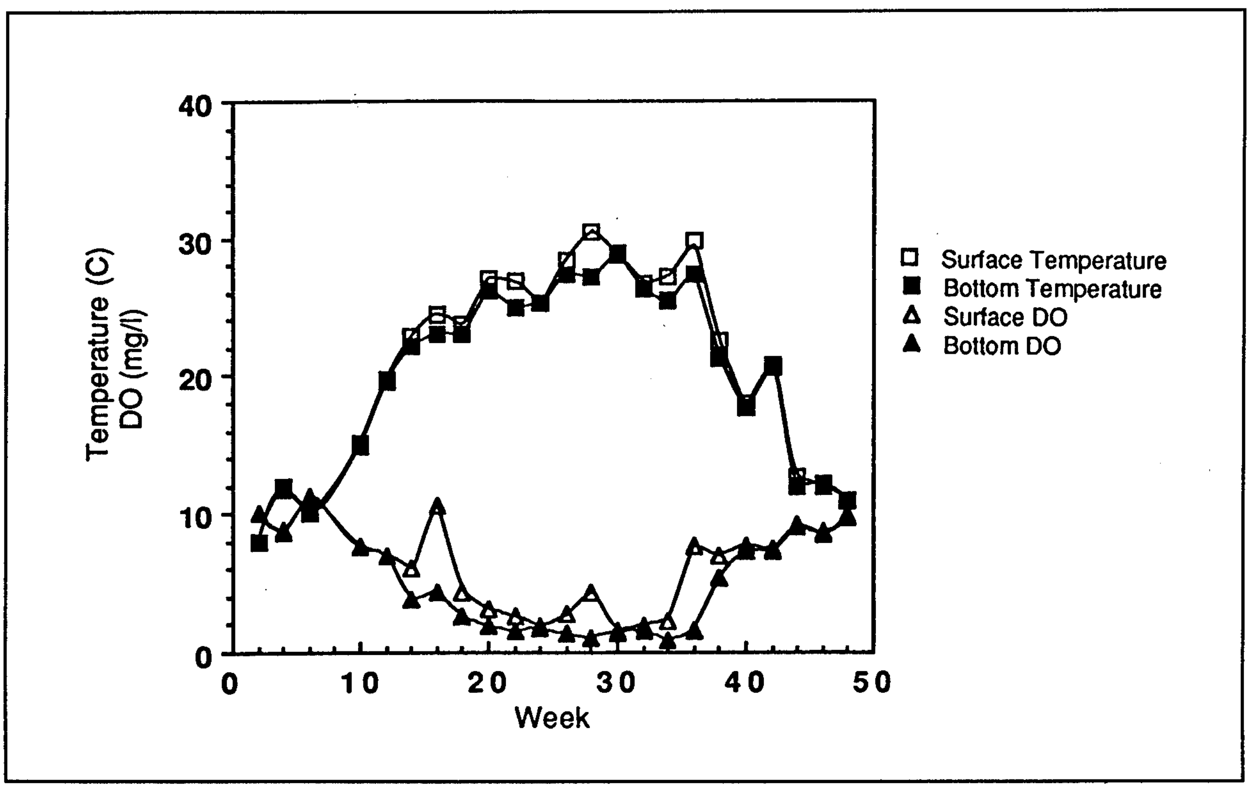

Figure 6. Surface and bottom temperature and DO levels at Elliot's Flats, Lake Marion, South Carolina, during 1991

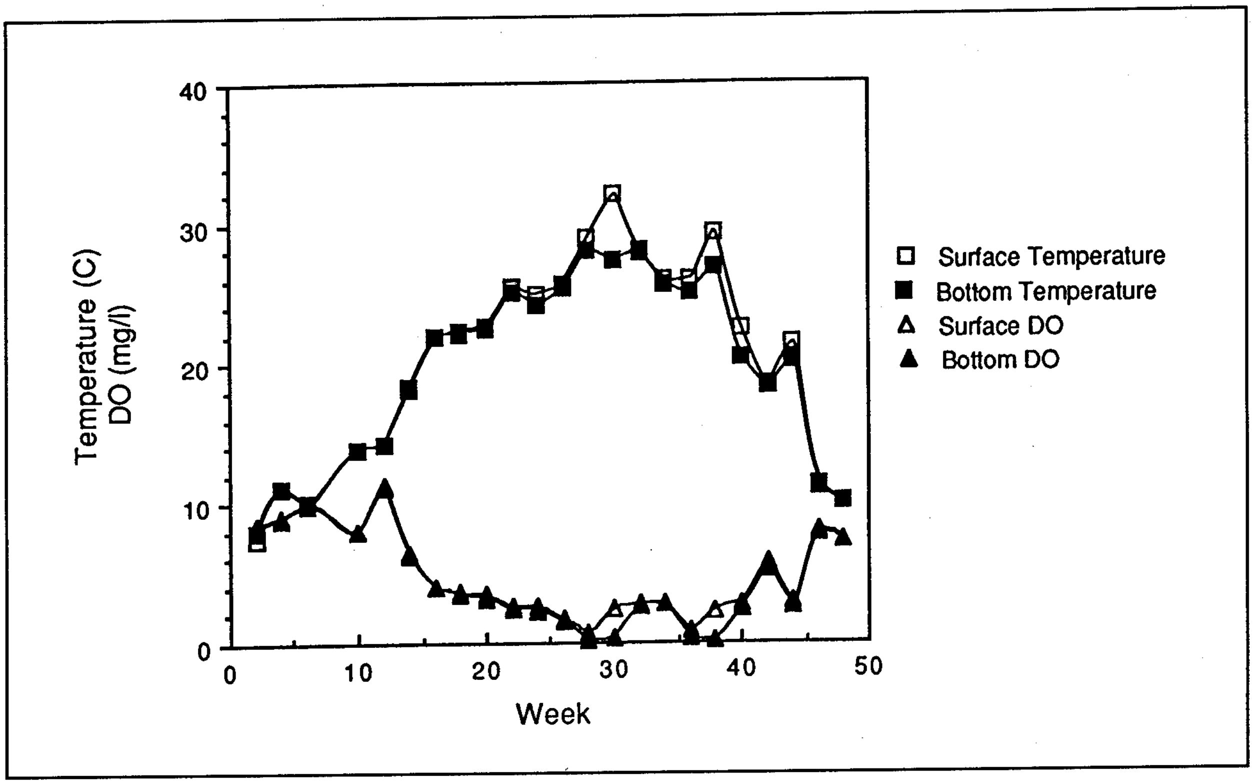

Figure 7. Surface and bottom temperature and DO levels at Sparkleberry Lake, Santee Swamp, South Carolina, during 1991

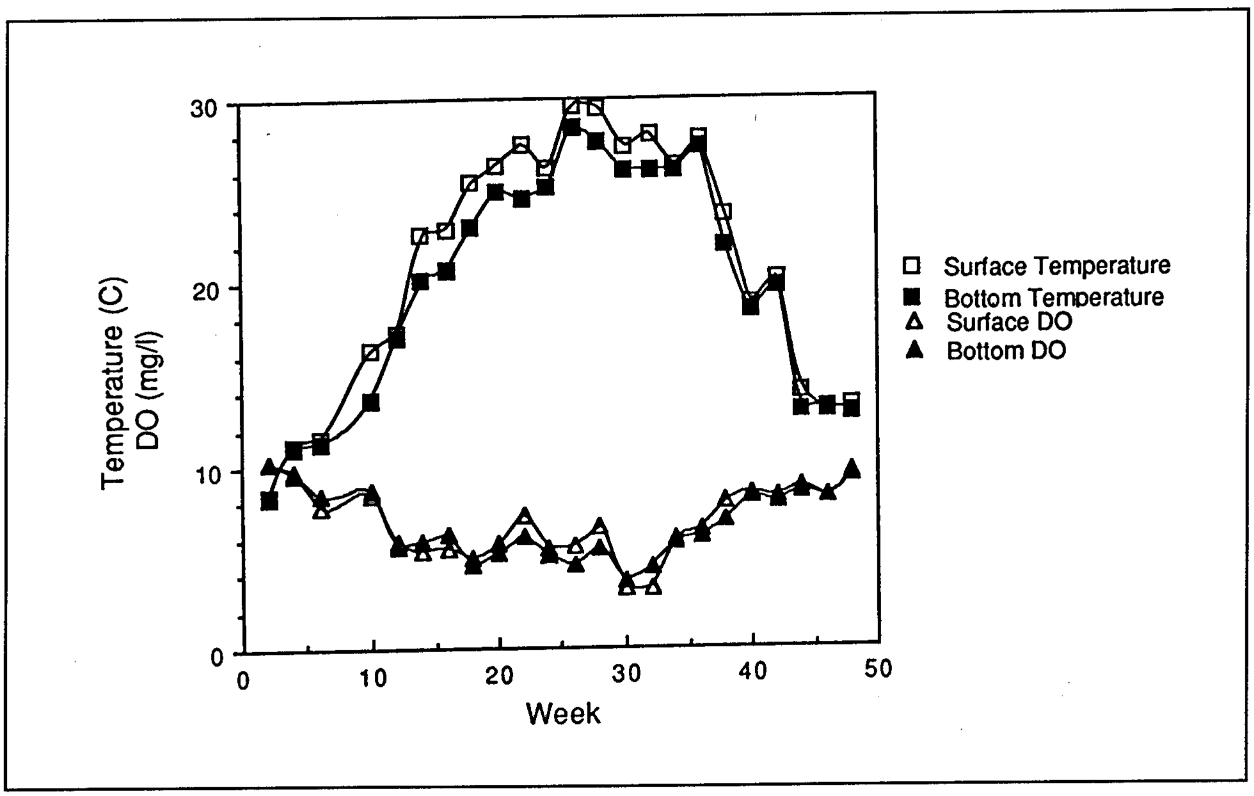

Figure 8. Surface and bottom temperature and DO levels at Brown's Lake, Lake Marion, South Carolina, during 1991

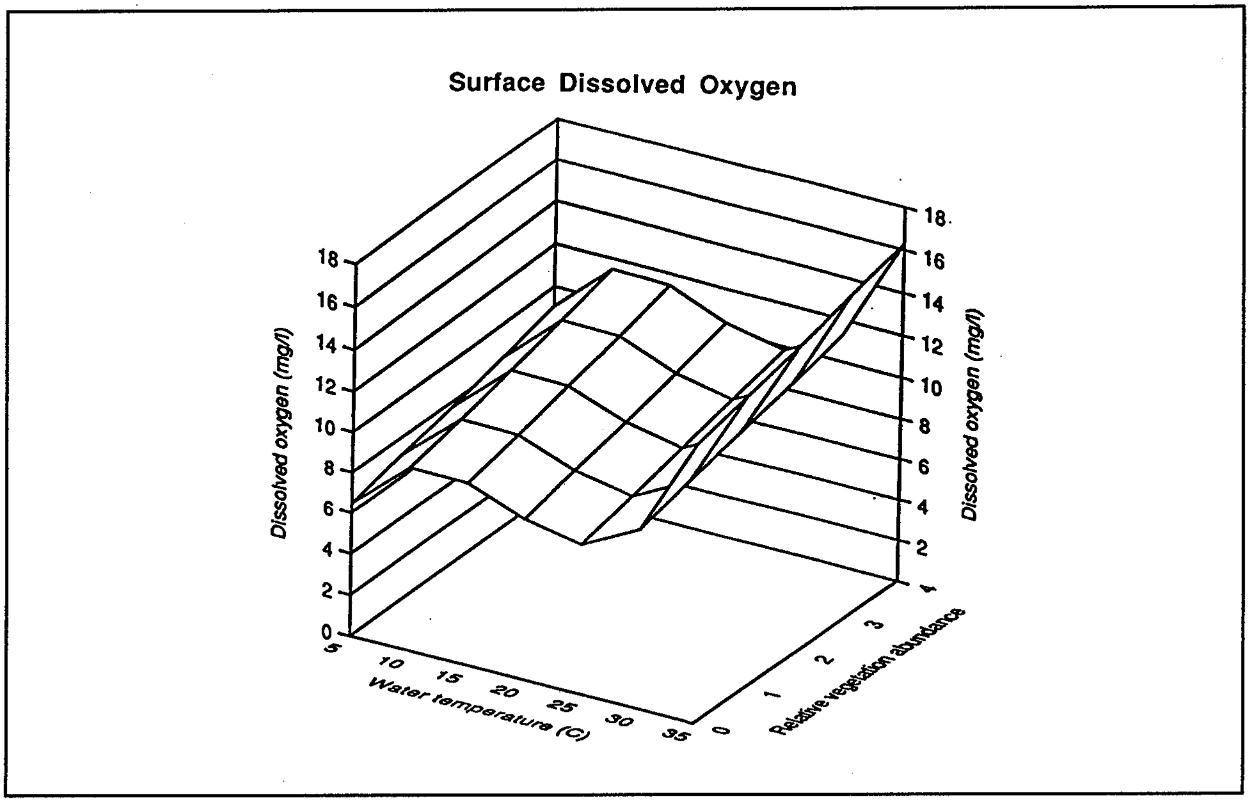

*Figure 9. Projected surface DO levels for four study areas in upper Lake Marion, South Carolina, during 1991. Surface DO = 1.786 * Surface temperature + 0.646 * Vegetation abundance - 0.115 * Surface temperature^2 + 0.002 * Surface temperature^3*

$p \leq 0.0001$) except Pack's Flats and Sparkleberry Lake ($p = 0.43$) and differed significantly between sampling periods ($F = 200.77$, $p \leq 0.0001$). The highest mean bottom DO level recorded was 11.3 mg/L at Pack's Flats on 19 February (week 6). The lowest mean bottom DO level recorded was 0.2 mg/L at Sparkleberry Lake on 11 July (week 26) and 17 September (week 36) and at Pack's Flats on 18 September (week 36). Bottom DO was related to bottom temperature and vegetation abundance (R-square = 0.93) (Figure 10). Bottom DO concentrations were relatively high at winter water temperatures and declined with increasing temperature, much like surface DO concentrations. Bottom DO concentrations continued to decline as vegetation abundance and bottom temperature increased.

Biweekly mean surface and bottom water temperatures were lowest during January. Temperatures increased throughout the year to a peak in July, August, and early September (weeks 26-36, Figures 4-8). After the summer peak, temperatures decreased throughout the remainder of the study period. Surface temperature differed significantly from bottom temperature for all areas ($t_{\text{Brown's Lake}} = 14.25$, $p \leq 0.0001$; $t_{\text{Elliot's Flats}} = 11.47$, $p \leq 0.0001$; $t_{\text{Pack's Flats}} = 12.20$, $p \leq 0.0001$; $t_{\text{Sparkleberry Lake}} = 8.04$, $p \leq 0.0001$; $t_{\text{Santee River}} = 6.29$, $p \leq 0.0001$).

Surface temperature differed significantly among all study areas ($F = 81.74$, $p \leq 0.0001$) except Brown's Lake (Figure 8) and Elliot's Flats ($p = 0.27$) and differed significantly among biweekly sampling periods ($F = 966.97$, $p \leq 0.0001$). The highest mean surface temperatures were recorded at Pack's Flats; 32.2 °C on 28 May (week 20); 32.5 °C on 11 July (week 26); 32.8 °C on 24 July (week 28) and 32.7 °C on 18 September (week 36). Lowest mean surface temperature recorded was 7.4 °C on 22 January (week 2) at Sparkleberry Lake. Pack's Flats and Santee River usually had the

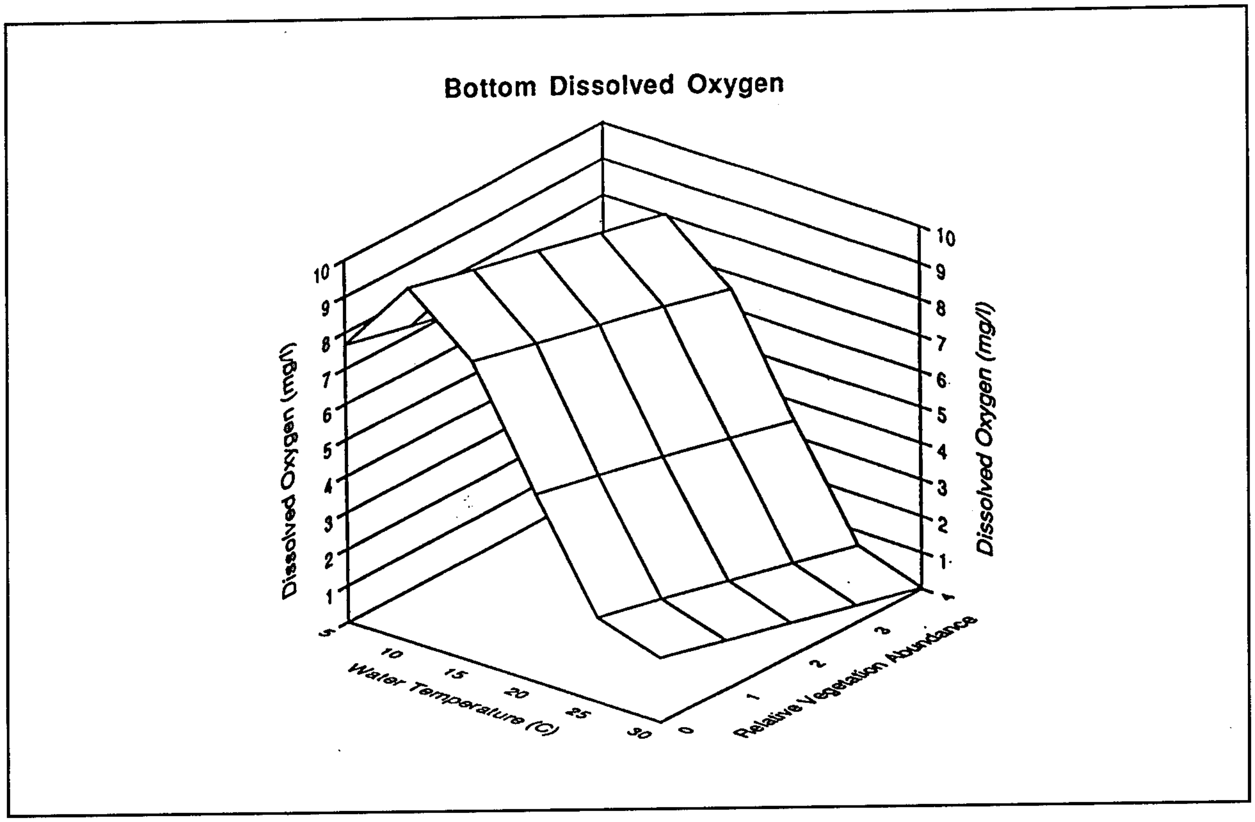

*Figure 10. Projected bottom DO levels for four study areas in upper Lake Marion, South Carolina, during 1991. Bottom DO = 2.224 * Bottom temperature - 0.448 * Vegetation abundance - 0.149 * Bottom temperature*^{2} *+ 0.003 * Bottom temperature*^{3}

highest and lowest surface temperatures, respectively.

Bottom temperatures were usually lower than surface temperatures. Santee River area usually had the lowest bottom temperatures of all study areas and showed the greatest fluctuations in bottom temperatures. Bottom temperature differed significantly among all study areas ($F = 13.33$, $p < 0.0001$) except Pack's Flats and Sparkleberry Lake ($p = 0.95$) and differed significantly between biweekly sampling periods ($F = 2015.89$, $p < 0.0001$). Highest mean bottom temperatures recorded were 28.8 °C at Pack's Flats and Elliot's Flats on 07 August (week 30) and 28.4 °C at Brown's Lake on 10 July (week 22). Lowest mean bottom temperatures recorded were 7.8 °C at Pack's Flats on 22 January (week 2) and 7.9 °C at Sparkleberry Lake on 23 January (week 2).

DO profiles and temperature profiles recorded on 17 September (week 36) showed the extent to which vegetation abundance influences the DO concentrations and temperature in the water column (Figure 11). Brown's Lake, which had the lowest vegetation abundance values, had uniform DO concentrations (6.5 mg/L at surface and 6.1 mg/L at bottom) throughout the water column. Temperatures at Brown's Lake were also uniform throughout the water column (27.8 °C at surface and 27.6 °C at bottom). In contrast, Pack's Flats, which had the highest vegetation abundance values, exhibited DO and temperature stratification. DO concentration and temperature declined from the surface (9.2 mg/L and 34.5 °C, respectively) to a depth of 30 cm (2.8 mg/L and 28.4 °C) and DO was very low (<1.0 mg/L) at the bottom. Elliot's Flats had relatively high vegetation abundance values and exhibited DO and temperature stratification, although not as extreme as Pack's Flats. Sparkleberry Lake had vegetation abundance values slightly higher than Brown's Lake and exhibited some stratification, but not nearly so much as Pack's Flats or Elliot's Flats. Triploid grass

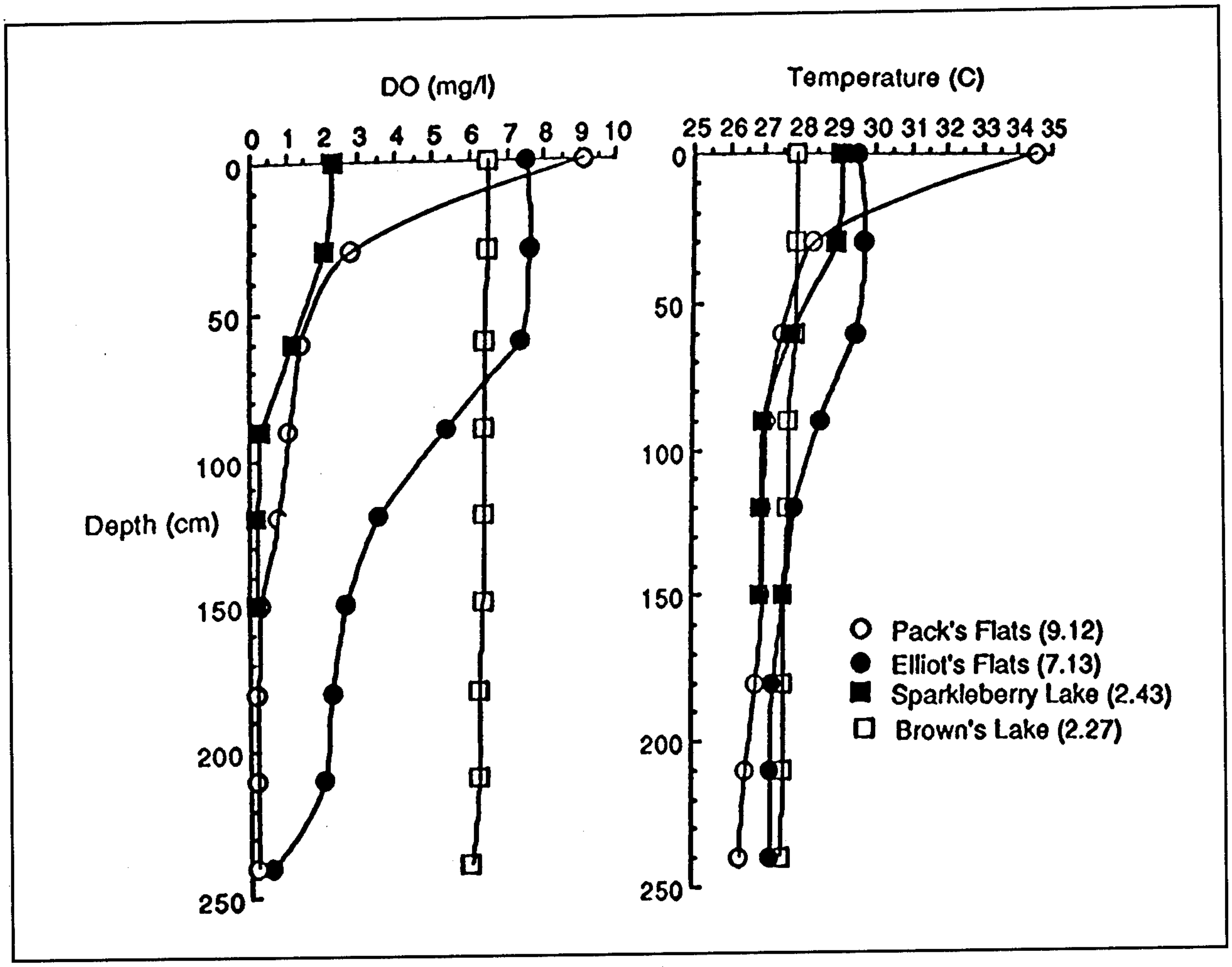

Figure 11. Temperature and DO profiles recorded on 17 September 1991 at four study areas in upper Lake Marion, South Carolina. Mean annual relative aquatic vegetation abundances reported in Figure 2 appear in parentheses

carp had achieved complete or near-complete control of aquatic vegetation in Elliot's Flats and Pack's Flats, respectively, by the summer of 1992. Temperature and DO profiles taken on 20 July 1992 and 29 July 1993 typify present summer conditions (Figures 12 and 13).

Discussion

Aquatic vegetation is a major problem for navigation and recreational activities in Lake Marion north of the I-95 bridge (Inabinette 1985). *Hydrilla* has been the most problematic species in Lake Marion in recent years. *Hydrilla* was the dominant vegetation species present in upper Lake Marion over the course of this study, regardless of season or water temperature. *Hydrilla* appeared to grow faster and form a surface canopy earlier in the year than any other species. This resulted in extensive, dense canopies of *Hydrilla* that shaded out other species of aquatic plants. Profound water quality impacts may occur when aquatic vegetation forms surface canopies over aquatic habitats, as *Hydrilla* did in upper Lake Marion. Buscemi (1958), Ultsch (1973), Schreiner (1980), and Frodge, Thomas, and Pauley (1990) described localized changes in water quality in ponds and lakes with patches of aquatic vegetation. Water quality characteristics in, under, or near plant beds was notably different from water quality characteristics in open-water areas. Physical and chemical characteristics of a vegetated area became more like characteristics of an open-water area when surface canopies of aquatic vegetation were physically removed (Frodge, Thomas, and Pauley 1990). Similar comparisons (between

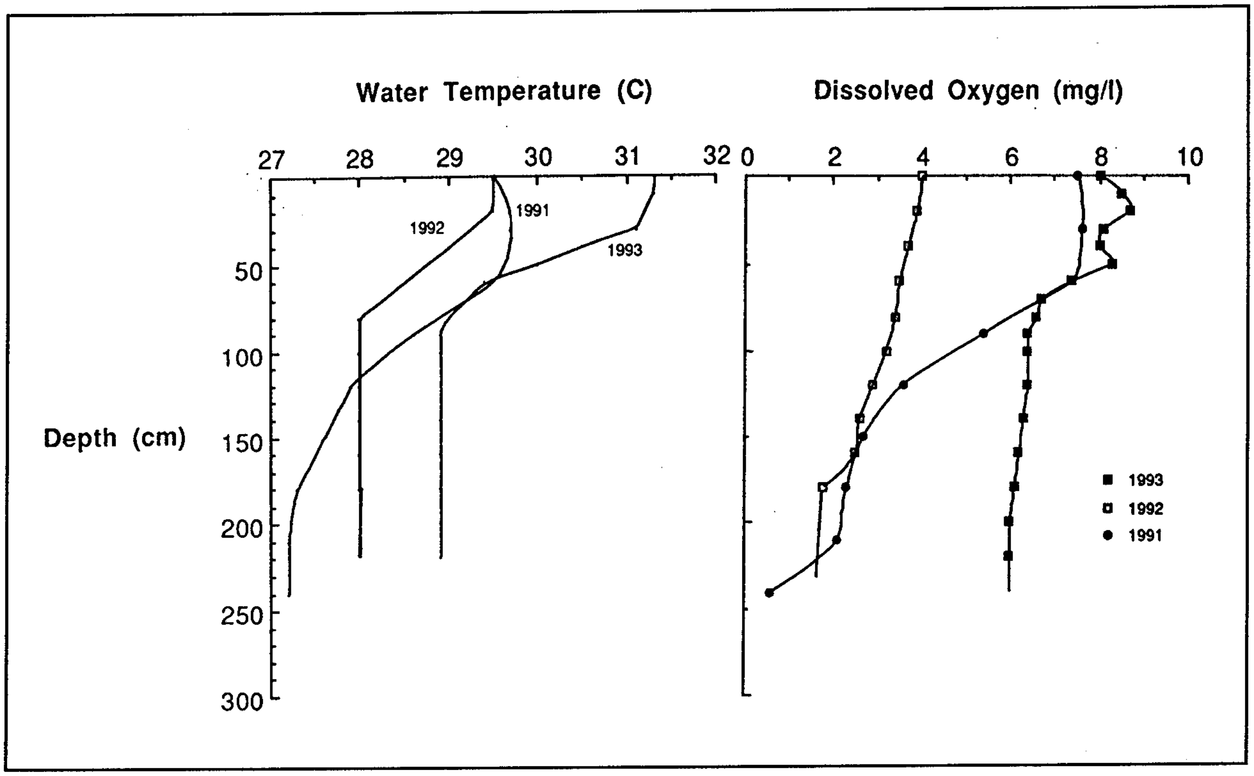

Figure 12. Summer temperature and DO profiles recorded at Elliot's Flats, upper Lake Marion, South Carolina

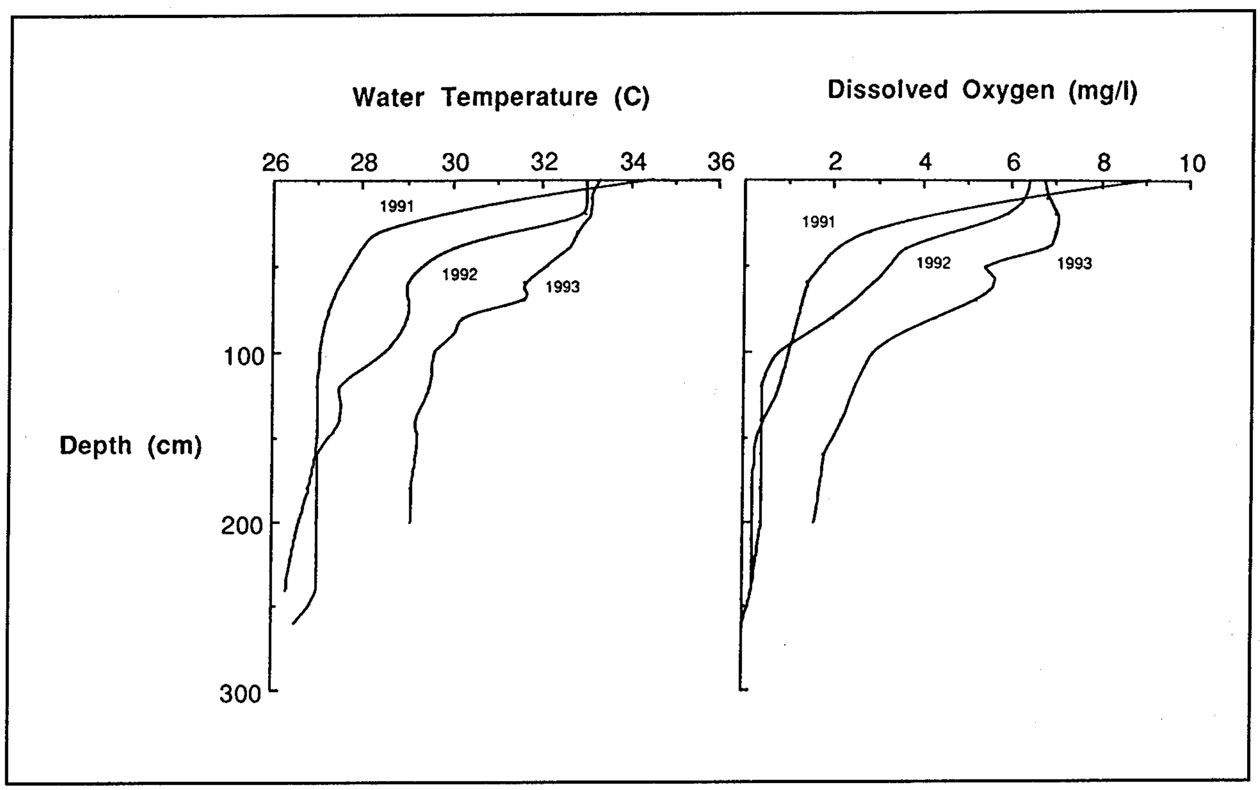

Figure 13. Summer temperature and DO profiles recorded at Pack's Flats, upper Lake Marion, South Carolina

vegetated and open-water areas) can be made in upper Lake Marion when heavily vegetated areas such as Pack's Flats are contrasted with less-infested areas such as Brown's Lake.

Areas of Lake Marion with dense growth of aquatic macrophytes (e.g., Pack's Flats and Elliot's Flats) exhibited distinctly different thermal characteristics from Brown's Lake, Sparkleberry Lake, and Santee River, where vegetation was sparse or absent. However, in winter when plant biomass was relatively low, thermal characteristics were similar for all study areas. Surface temperatures in vegetation canopies were very high (>30 °C) in Pack's Flats and Elliot's Flats during the summer months when canopies became extensive. Bottom temperatures were usually 5 to 10 °C lower than surface temperatures in these areas. In contrast, temperatures were more uniform throughout the water column at Brown's Lake, Sparkleberry Lake, and Santee River, where extensive surface canopies did not form. The extent of thermal stratification was apparently related to vegetation abundance. Thermal stratification was reported present in plant beds and absent in open-water areas by Rorslett, Berge, and Johansen (1986), Wylie and Jones (1987), and Frodge, Thomas, and Pauley (1990).

DO concentrations differed greatly from location to location in Pack's Flats and Elliot's Flats when dense surface canopies were present. This was probably because of limited horizontal mixing through dense growths of aquatic vegetation. DO concentrations as high as 15.0 mg/L were recorded in surface canopies. These supersaturated DO levels were similar to levels reported by Rorslett, Berge, and Johansen (1986), Wylie and Jones (1987) and Frodge, Thomas, and Pauley (1990). Supersaturation is apparently caused by photosynthetic activity by surface vegetation. Surface DO concentrations were related to vegetation abundance and water temperature (Figure 9). Winter surface DO levels were relatively low. Even though oxygen solubility is relatively high at winter water temperatures, photosynthetic activity by aquatic macrophytes and phytoplankton is low at winter temperatures and short photoperiod durations. The slight increase in surface DO levels as water temperatures increased from 5 to 10 °C was probably because of increased phytoplankton and macrophyte photosynthesis. In spring and early summer as water temperatures increased from 10 to 25 °C, the increased photosynthetic activity of macrophytes and phytoplankton was probably more than offset by the decreasing solubility of oxygen. As water temperatures increased above 25 °C and relative vegetation abundance increased, photosynthetic activity probably exceeded the effect of decreased oxygen solubility.

Bottom DO levels were lower beneath dense canopies of vegetation. Bottom DO levels were related to vegetation abundance and bottom temperatures (Figure 10). As water temperatures increased, oxygen solubility in water decreased. As vegetation abundance increased, the surface canopy became more dense, allowing less light to reach the bottom and thus decreasing photosynthetic activity at the bottom. Also, as the surface canopy becomes vertically thicker and more dense, some of the bottom layer of the surface canopy will die and fall to the bottom, thus increasing biological oxygen demand at or near the bottom. Subsurface oxygen depletion beneath canopies of aquatic macrophytes has been reported by Buscemi (1958), Wylie and Jones (1987), and Frodge, Thomas, and Pauley (1990).

The water quality conditions present in Elliot's Flats, Pack's Flats, and Sparkleberry Lake during summer present a problem for triploid grass carp. Although grass carp have low oxygen requirements (Opuszynski 1972), conditions present in these areas in the summer is probably not favorable for maximum feeding and growth rates. Chappelear (1990) reported a general downlake movement of triploid grass carp in upper Lake Marion during the summer of 1989. He hypothesized that this movement was due to low DO levels present in the uppermost areas of Lake Marion during the summer time. Triploid grass carp stocked downlake of Pack's Flats during 1990 generally remained in this area, with

only a small percentage entering Pack's Flats (Kartalia 1992). Areas such as Pack's Flats, where water quality conditions are most extreme during the summer, do not provide suitable habitat for grass carp in the summer if a complete *Hydrilla* canopy exists. However, grass carp may enter these areas in fall, winter, and spring when water quality conditions are more favorable because aquatic vegetation is present year round. Complete or near-complete *Hydrilla* control and removal of a *Hydrilla* canopy has resulted in temperature and DO levels acceptable to triploid grass carp and native fishes.

There is no general consensus on the impact that reduction of aquatic macrophytes by grass carp has on fish communities (Carpenter and Lodge 1986). Removal of all aquatic vegetation in upper Lake Marion would reduce habitat variability and have an adverse effect on the fish community. However, this study indicates that a reduction of aquatic plant biomass in upper Lake Marion, in addition to improving recreational access, would improve water quality in Lake Marion. As a result of improved water quality, many areas would then be capable of supporting complex fish communities.

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Florida Case Histories

Use of Grass Carp in Two Florida Lakes, 1975 to 1994

by

Douglas E. Colle[1] *and Jerome V. Shireman*[1]

Introduction

The multirecreational usage of water bodies is continually increasing throughout the country. As user groups become more diverse in their recreational pursuits, which include sailing, windsurfing, water-skiing, swimming, and boating, their requests for aquatic macrophyte control are steadily increasing. This increased demand coupled with the introduction of the exotic aquatic macrophyte hydrilla (*Hydrilla verticillata*) into the United States has generated considerable interest in the development of biological plant controls to alleviate the continued usage of aquatic herbicides.

The grass carp (*Ctenopharyngodon idella*) was first introduced into the United States for plant control in Arkansas and Alabama in 1963 (Guillory and Gasaway 1978). The use of grass carp for vegetation control has been extremely controversial. The two major concerns have been fear of reproduction and the potential negative impact that total aquatic macrophyte elimination might have upon aquatic systems. The reproduction issue has been addressed with the commercial production of sterile grass carp (Allen and Wattendorf 1987). Long-term (>10 years) submersed vegetation elimination has been documented for grass carp (Leslie et al. 1987). The removal of vegetation has resulted in increased total phosphorous, total nitrogen, chlorophyll *a* values, and turbidity and a decrease in Secchi disk transparency (Leslie, Nall, and Van Dyke 1983; Shireman and Hoyer 1986). The majority of reported fisheries investigations have had 1 to 2 years of background data and from 2 to 4 years of results after submersed vegetation removal by grass carp (Ware and Gasaway 1976; Bailey 1978; Shireman and Hoyer 1986; Klussmann et al. 1988). This report summarizes 14 years of research in two Florida lakes using grass carp for aquatic macrophyte control.

Study Sites

Lake Baldwin is an 80-ha urban lake located within the United States Naval Training Center in Orlando, Orange County, Florida. The lake has a maximum depth of 7.8 m and a mean depth of 4.4 m. Water tupelo (*Nyssa aquatica*) occupies 35 percent of the shoreline extending out to a depth of 0.75 m. Hydrilla has been the dominant submersed plant in Lake Baldwin since 1971, covering as much as 80 percent of the lake bottom; other submersed macrophytes are southern naiad (*Najas quadalupensis*) and eel-grass (*Vallisneria americana*). Cattail (*Typha latifolia*), maidencane (*Panicum hemitomon*), and torpedograss (*Panicum repens*) are present along the shoreline.

Lake Pearl is a 24-ha, urban lake located in Orange County, Florida. Urban development was present on 40 percent of Lake Pearl's shoreline in 1978; by 1990, most of the shoreline was developed. The lake has a maximum depth of 7.0 m and a mean depth of 2.5 m. Hydrilla was first documented in Lake Pearl in 1977, and by 1978, it covered 95 percent of the lake bottom. Surface coverage of native rooted aquatic vegetation at project initiation was approximately 10 percent; 5 percent in a 12-m shoreline band of nutgrass (*Sceleria* sp.) and 5 percent consisting of interspersed beds of spatterdock (*Nuphar luteum*) and fragrant water lily (*Nymphaea odorata*). Small communities of cattail, torpedo grass, and pickerelweed (*Pontederia cordata*) totaled less than 1-percent surface coverage. Bladderwort

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(*Utricularia subulata* and *U. inflata*) covered 14 percent of the lake surface area in association with the hydrilla community.

Methods

Lake Baldwin submersed vegetation was treated with aquatic herbicides from 1971 to 1978; but after 1978, no aquatic herbicides were applied to the lake. Diploid grass carp were initially stocked in April 1975 with 4,999 (63/hectare) fish (mean length 144-mm total length (TL)). By 1977, hydrilla covered 35 percent of the lake bottom; therefore, a selective rotenone treatment was conducted in October 1977 to obtain a grass carp population estimate (Colle et al. 1978). Only 264 (3/hectare) grass carp (169 to 434, 95-percent confidence intervals) remained from the initial stock. The lake was restocked during the summer of 1978 with an additional 1,865 (23/hectare) larger grass carp (>304-mm TL) to reduce the likelihood of predation by largemouth bass (Shireman, Colle, and Canfield 1978).

Fifty percent of Lake Pearl submersed vegetation was treated with aquatic herbicides from March 1980 through February 1982; herbicides have not been applied since 1982. The lake was stocked with diploid grass carp (>300-mm TL) in July 1980 (143 fish), July 1981 (284 fish), December 1981 (300 fish), and March 1982 (300 fish) for a total of 1,027 grass carp (43/hectare).

A recording fathometer was used to determine the percent of bottom coverage and percent of the lake volume containing submersed vegetation (PVI). Water samples were collected 0.5 m below the surface at midlake stations. Water quality analysis was conducted by the University of Florida utilizing standard methods (American Public Health Association 1992) for chlorophyll *a*. Total nitrogen concentrations were determined using Nelson and Sommers' (1975) modified Kjeldahl technique. Total phosphorous concentrations were analyzed by Murphy and Riley's (1962) procedures with a persulfate digestion (Menzel and Corwin 1965). Secchi disk transparency was measured using a standard black and white Secchi disk.

Annual fall fish population estimates were conducted in Lakes Baldwin and Pearl with block-nets (0.08 ha) and rotenone following procedures outlined by Shireman, Colle, and Durant (1981); a minimum of three nets were set in all study years. Estimates of the Florida largemouth bass (*Micropterus salmoides floridanus*) population in Lake Pearl were calculated from electrofishing samples with Chapman's modification of the Schnabel method; confidence intervals were determined from Ricker (1975). Estimates were determined for subadult (151 to 250-mm TL), adult (>250-mm TL), and large female (>500-mm TL) largemouth bass. All fish were weighed, measured, and tagged with numbered Floy FD-68BC tags. Growth allowances were made from the tag data to prevent recruitment bias during the estimate period (4 to 8 months). Adult (≥160-mm TL) bluegill (*Lepomis macrochirus*) and redear sunfish populations were estimated in 1987 with Chapman's adjusted Peterson estimate; confidence intervals were determined from Ricker (1975). Fish were collected with electrofishing gear, and a pelvic fin was clipped for an identifiable mark.

Results

Aquatic vegetation

The 1975 grass carp stock had no impact upon hydrilla coverage in Lake Baldwin (Table 1). During 1977, the mean individual biomass of grass carp was 12.23 kg, with a standing stock estimate of three fish and 40.36 kg/hectare (Colle et al. 1978). However, the second grass carp stocking in 1978 resulted in hydrilla elimination by June of 1980. The plant has not been collected in Lake Baldwin as of April 1994 (Table 1). Grass carp averaged 9.17 kg during June 1980, with an estimated standing stock of 26 fish/hectare and 235 kg/hectare (Shireman and Maceina 1981). Eel-grass, southern naiad, maidencane, torpedograss, cattail, and

Table 1
Water Quality Parameters, Percent Submersed Vegetation Coverage and PVI for Lake Baldwin, Florida

Year	Percent Cover	PVI	Secchi Depth m	Total Phosphorus mg/m^4	Total Nitrogen mg/m^4	Chlorophyll *a* mg/m^4
1977	35	–	3.1	17.5	465	5.9
1978	69	37	5.0	11.3	870	1.0
1979	57	19	1.5	45.7	832	26.1
1980	11	1	1.5	33.9	628	16.2
1981	0	0	1.3	33.5	694	24.7
1982	0	0	1.6	21.5	526	14.1
1983	0	0	1.8	23.9	462	10.6
1984	0	0	1.6	32.3	561	14.8
1985	0	0	1.5	19.0	445	9.0
1986	0	0	1.5	21.0	573	15.9
1987	0	0	1.4	27.5	635	19.1
1988	0	0	1.3	22.8	577	24.5
1989	0	0	1.6	15.3	270	13.5
1990	0	0	1.6	24.6	687	13.2
1993	0	0	1.3	19.0	726	20.3
1994	0	0	1.5	27.3	395	15.2

all other aquatic macrophytes were consumed by the winter of 1982, and regrowth has not occurred for any submersed macrophytes as of April 1994. During 1993, both cattail and panic grass communities began to revegetate along the lake shoreline. The blue-green alga (*Lyngbya* sp., averaging <15 cm high) covered 23 percent of the lake bottom in water depths to 3.5 m from 1985 through 1990. However, vegetation transects in 1993 and 1994 indicated only trace amounts of *Lyngbya* sp. within the lake. Lake Baldwin grass carp mean weights declined to 6.16 kg after 1985. Periodic food habits analysis showed grass carp were consuming filamentous algae and detritus after macrophyte elimination. Although not collected, grass carp were visually sighted in 1994 electrofishing samples in Lake Baldwin.

Lake Pearl received 50-percent areal treatments of aquatic herbicides from March 1980 through February 1982, coupled with grass carp stockings in an attempt to control hydrilla without totally eliminating all submersed macrophytes. The initial hypothesis was that grass carp would select hydrilla regrowth in the herbicided region; however, herbicide applications were required at approximately 2-month intervals until February 1982. This treatment schedule was comparable with other lakes in Orange County, Florida, where only aquatic herbicides were utilized for aquatic plant management. By August 1982, all submersed vegetation was eliminated, and no submersed macrophyte regrowth had occurred by April 1994 (Table 2). During the summer of 1982, when macrophytes were eliminated, grass carp averaged 4.57 kg; assuming a 62-percent survival rate (based on results from Shireman and Hoyer 1986), the standing stock estimate would be 27 fish/hectare and 123 kg/hectare. All emergent vegetation was consumed by the fall of 1983, and as of April 1994, the only shoreline vegetation was giant cutgrass (*Zizaniopsis miliacea*). The only aquatic vegetation still present is the spatterdock and fragrant water lily communities, which had 3-percent areal coverage as of April 1994.

Water quality

Complete removal of submersed vegetation produced significant long-term changes in certain water quality values in both study lakes, but the trophic status of the lakes did not change (Tables 1-2). Chlorophyll *a* values increased in both lakes after vegetation removal and have remained higher as the systems shifted from macrophytic- to phytophytic-dominated lakes. Lake Pearl chlorophyll *a* values were negatively correlated to both percent submersed-vegetation coverage ($p < 0.05$, $r = -0.66$) and PVI ($p < 0.05$, $r = -0.68$) and positively correlated with year ($p < 0.01$,

Table 2
Water Quality Parameters, Percent Submersed Vegetation Coverage and PVI for Lake Pearl, Florida

Year	Percent Cover	PVI	Secchi Depth m	Total Phosphorus mg/m^4	Total Nitrogen mg/m^4	Chlorophyll *a* mg/m^4
1978	89	63	–	–	–	–
1979	89	63	2.2	–	–	9
1980	93	64	2.2	20.2	783	4
1981	90	54	2.2	20.3	578	13
1982	35	17	0.8	21.8	703	24
1983	0	0	1.2	28.8	881	13
1984	0	0	1.0	24.5	862	17
1985	0	0	1.0	28.5	923	23
1986	0	0	0.9	33.2	892	20
1987	0	0	0.9	32.4	949	25
1988	0	0	0.9	36.5	695	24
1989	0	0	0.9	28.2	820	25
1990	0	0	1.0	32.1	897	44
1994	0	0	1.0	27.4	–	23

$r = +0.72$). Lake Baldwin chlorophyll *a* values were not significantly correlated to vegetation coverage, PVI, or year. Concurrently, Secchi disk transparency was reduced at least 50 percent in both lakes with increased algal biomass as measured by chlorophyll *a* concentrations. Secchi disk transparencies were positively correlated with plant coverage and PVI and negatively correlated with year in both systems (Baldwin: $p < 0.01$, percent cover 0.76, PVI 0.87, year $p < 0.05$, $r = -0.51$; Pearl: $p < 0.01$, percent cover 0.93, PVI 0.95, year -0.68). Total nitrogen values in Lake Baldwin were positively correlated with both percent cover ($p < 0.5$, $r = 0.57$) and PVI ($p < 0.05$, $r = 0.64$) and not significantly correlated to year. Lake Pearl total nitrogen was negatively correlated to both percent cover ($p < 0.05$, $r = -0.67$) and PVI ($p < 0.05$, $r = -0.62$) and not significantly correlated with year. Lake Baldwin total phosphorous values were not correlated to either percent plant cover, PVI, or year. Total phosphorous levels in Lake Pearl were negatively correlated to both percent cover ($p < 0.01$, $r = -0.78$) and PVI ($p < 0.01$, $r = -0.76$) and positively correlated with year ($p < 0.01$, $r = 0.82$).

Fish populations

Total number and biomass of fish in Lake Baldwin were not significantly correlated to percent macrophyte coverage, PVI, or year. Mean number of fish per hectare in Lake Baldwin were equivalent during the vegetated and nonvegetated years (Table 3). Total kilograms per hectare were also equivalent when submersed vegetation was present and when Lake Baldwin was devoid of submersed macrophytes. Estimates of total number of fish per hectare in Lake Pearl were positively correlated with plant coverage and PVI and negatively correlated with year (Table 4). Mean number of fish per hectare when Lake Pearl had submersed vegetation was greater than in the nonvegetated years (Table 5). Biomass estimates in Lake Pearl were positively correlated to percent plant coverage and PVI, but were not significantly correlated with year. Mean biomass estimates during years with submersed vegetation were greater than in years when Lake Pearl had no submersed plants (Table 5).

The elimination of submersed vegetation in Lakes Baldwin and Pearl resulted in significant population declines in both systems for the following species: lake chubsucker (*Erimyzon sucetta*), golden shiners (*Notemigonus crysoleucas*), and warmouth (*Lepomis gulosus*). The loss of submersed vegetation in Lake Pearl resulted in the elimination or reduction in the populations to levels below detection for chain pickerel (*Esox niger*), redfin pickerel (*Esox americanus*), taillight shiner (*Notropis maculatus*), golden topminnow (*Fundulus chrysotus*), bluespotted sunfish (*Enneacanthus gloriosus*), and Everglades pygmy sunfish

Table 3
Annual Fish Population Standing Crop Estimates (number and kilograms per hectare) from Block-net and Rotenone Samples in Lake Baldwin, Florida

Year	Total		Bluegill (<160-mmTL)		Bluegill (≥160-mmTL)		Redear (<160-mmTL)		Redear (≥160-mmTL)		Largemouth Bass (<240-mmTL)		Largemouth Bass (≥240-mmTL)	
	Number	Kilograms	Number	Kilograms	Number	Kilograms	Number	Kilograms	Number	Kilograms	Number	Kilograms	Number	Kilograms
1977	16,223	141	6,134	23.0	74	2.9	1,170	7.4	31	3.3	386	3.6	14	4.5
1978	—	—	—	—	—	—	—	—	—	—	—	—	—	—
1979	8,049	199	4,532	48.7	9	0.6	942	16.0	47	5.5	33	0.3	35	6.4
1980	11,028	54	2,849	2.1	1	0.1	5	0.1	1	0.1	0	0	25	38.1
1981	35,950	102	2,104	12.8	8	0.5	6	0.1	7	0.5	52	0.2	11	7.1
1982	12,924	75	7,357	17.2	115	12.8	91	0.2	139	7.6	8	0.1	20	4.1
1983	5,497	28	418	0.7	24	0.6	122	0.8	31	3.3	4	0.8	22	7.8
1984	9,497	69	1,912	4.9	156	12.4	659	7.7	53	3.7	123	0.6	21	3.6
1985	13,675	206	5,512	73.7	172	11.9	1,019	21.9	136	11.6	135	3.1	32	5.1
1986	14,390	143	4,670	74.5	152	11.6	576	16.5	81	8.9	137	1.7	30	6.1
1987	—	—	—	—	174[a]	13.1[a]	—	—	156[a]	15.9[a]	—	—	19	8.2
1988	7,104	79	4,467	37.1	22	1.0	358	6.7	8	10.7	92	0.3	10	2.4
1989	8,371	101	3,799	42.6	33	2.6	167	3.4	35	2.8	48	0.4	27	20.9
1977-1980[b]	11,767 (2,388)	131 (42)	4,505 (948)	24.6 (13.5)	28 (23)	1.2 (0.9)	706 (356)	7.8 (4.8)	26 (13)	2.9 (1.6)	140 (124)	1.3 (1.1)	25 (6)	16.3 (10.9)
1981-1989[b]	12,108 (3,286)	91 (19)	3,780 (788)	32.9 (10.3)	95 (24)	7.4 (2.0)	375 (124)	(7.7) (2.9)	72 (20)	7.2 (1.7)	75 (19)	0.9 (0.4)	21 (3)	7.3 (1.8)

[a] Mark-recapture estimates.
[b] Mean of annual means and (standard error).

Table 4
Plant Coverage, PVI, and Year Correlation Coefficients for Standing Crops per Hectare of Fish Population Estimates in Lake Pearl, Florida

Species	Plant Coverage		PVI		Year	
	p<	r	p<	r	p<	r
Total number	0.01	0.75	0.01	0.78	0.05	-0.61
Total kilograms	0.05	0.70	0.05	0.71	—	—
Bluegill <160, kg	0.01	0.73	0.01	0.71	—	—
Redear ≥160, number	0.05	-0.63	0.05	-0.62	0.01	0.74
Redear >160, kg	0.05	-0.65	0.05	-0.63	0.01	0.80
Largemouth bass <160, number	0.05	0.69	0.05	0.66	0.05	-0.57
Largemouth bass ≥250, number	0.05	0.62	0.05	0.55	0.05	-0.67

Table 5
Annual Fish Population Standing Crop Estimates (number and kilograms per hectare) from Block-net and Rotenone Samples in Lake Pearl, Florida

Year	Total		Bluegill (<160-mmTL)		Bluegill (≥160-mmTL)		Redear (<160-mmTL)		Redear (≥160-mmTL)		Largemouth Bass (<160-mmTL)
	Number	Kilograms	Number	Kilograms	Number	Kilograms	Number	Kilograms	Number	Kilograms	Number
1978	33,868	130	2,645	25.2	41	2.8	342	2.2	8	0.2	97
1979	10,862	254	2,815	68.8	248	21.2	188	4.4	9	0.2	97
1980	30,617	118	22,427	44.0	17	1.5	858	3.5	8	0.1	68
1981	19,547	183	10,638	84.6	86	6.4	541	6.7	3	0.1	217
1982	1,584	63	884	11.7	10	0.9	86	2.9	35	2.4	4
1983	7,852	41	5,944	10.5	54	4.5	752	1.5	4	0.1	44
1984	8,044	45	5,697	15.7	42	4.3	626	3.0	18	2.7	36
1985	5,979	89	3,856	17.3	5	0.5	347	5.1	179	22.3	58
1986	8,924	58	6,322	31.2	22	3.0	129	1.6	205	29.5	49
1987	—	—	—	—	—	—	—	—	—	—	—
1988	9,472	123	1,981	16.7	51	5.1	1,132	3.0	92	14.0	5
1989	8,051	118	3,062	22.8	184	25.0	278	2.0	145	25.0	10
1978-1982[a]	19,295 (6,022)	150 (32)	7,881 (4,007)	46.9 (13.5)	80 (43)	6.5 (3.8)	403 (137)	3.9 (0.8)	13 (6)	0.6 (0.5)	97 (35)
1983-1989[a]	8,053 (487)	79 (15)	4,477 (723)	19.0 (2.9)	60 (26)	7.1 (3.7)	544 (150)	2.7 (0.6)	107 (34)	15.6 (4.6)	34 (9)

[a] Mean of annual means and (standard error).

(*Elassoma evergladei*). Gizzard shad (*Dorosoma cepedianum*) and threadfin shad (*Dorosoma petenense*) abundance increased significantly in Lake Pearl with the elimination of submersed vegetation. The Lake Baldwin threadfin shad population also increased significantly after the loss of submersed macrophytes.

Subadult bluegill (<160-mm TL) number and kilogram/hectare estimates in Lake Baldwin were not significantly correlated to percent plant coverage, PVI, or year. Lake Baldwin subadult bluegill standing crop estimates were equivalent in both vegetated and nonvegetated years (Table 3). Adult bluegill (≥160-mm TL) were not significantly correlated to percent plant coverage, PVI, or year. Although mean standing crop estimates of adult bluegill in Lake Baldwin were lower during vegetated years than in nonvegetated years, the 95-percent confidence intervals overlapped. Subadult bluegill number/hectare in Lake Pearl were not significantly correlated to plant coverage, PVI, or year. However, kilogram/hectare estimates were positively correlated to percent plant coverage and PVI, but were not significantly correlated with year (Table 4). Adult bluegill standing crop estimates were not significantly correlated with plant coverage, PVI or year in Lake Pearl. Both number/hectare and kilograms/hectare estimates of adult bluegill were equivalent for both vegetated and nonvegetated years in Lake Pearl (Table 5).

Subadult (<160-mm TL) and adult (≥160-mm TL) redear sunfish standing crop estimates in Lake Baldwin were not significantly correlated to plant coverage, PVI, or year. Subadult redear sunfish numbers/hectare and kilograms/hectare estimates were equivalent in Lake Baldwin during vegetated and nonvegetated years (Table 3). Adult redear sunfish standing crop estimates were also equivalent in vegetated and nonvegetated years. Subadult redear sunfish populations in Lake Pearl were not significantly correlated with plant coverage, PVI, or year. Standing crop estimates for subadult redear sunfish in Lake Pearl were equivalent in vegetated and nonvegetated years (Table 5). Adult redear sunfish in Lake Pearl were negatively correlated with plant coverage and PVI and positively correlated with year. Lake Pearl adult redear sunfish populations were significantly higher in nonvegetated years than vegetated years.

Subadult (<240-mm TL) and adult (≥240-mm TL) largemouth bass populations in Lake Baldwin were not significantly correlated to percent plant coverage, PVI, or year. Subadult standing crop estimates were equivalent during vegetated and nonvegetated years (Table 3). Adult largemouth bass populations were not significantly different during years with and years without submersed plants. Young-of-year largemouth bass numbers/hectare (<160-mm TL) in Lake Pearl were positively correlated with plant coverage and PVI and negatively correlated with year (Table 4). Annual standing crop estimates for young-of-year largemouth bass were greater during years with submersed vegetation except 1982 when only four fish/hectare were collected; however, all submersed vegetation had been consumed by June 1982, and block-net and rotenone sampling was not conducted until October (Table 5).

Mark-recapture estimates for largemouth bass (151- to 250-mm TL) in Lake Pearl were not correlated with percent plant cover, PVI, or year. Population estimates of subadult bass peaked from 1981 to 1983, with equivalent annual estimates before and after this period (Table 6). Subadult largemouth bass numbers were not significantly correlated to the previous year's standing crop of young-of-year largemouth bass. Adult largemouth bass (≥250-mm TL) numbers were positively correlated with plant coverage and PVI and negatively correlated to year; however, biomass estimates were not significantly correlated (Table 4). Annual estimates of adult bass were larger during vegetated years, but 95-percent confidence intervals overlapped during almost all study years (Table 6). The mean of means of the annual estimates from 1979 to 1983 was 49 fish/hectare (90-percent confidence intervals: 39 to 59); while during the nonvegetated years (1984-1989), the mean of means was 25 fish/hectare (90-percent confidence intervals:

15 to 36). Annual mark-recapture estimates for largemouth ≥500-mm TL, which represents the population of fish that are at least 4 years old and primarily female, were not correlated with plant coverage, PVI, or year. Confidence intervals overlapped during all study years except 1981 when the annual estimate was greater than most study years (Table 6).

Annual growth rates of Lake Pearl largemouth bass were estimated from Floy tag data. Largemouth bass <350-mm TL had significantly faster growth rates during nonvegetated years (Table 7). However, the growth rates of largemouth bass ≥350-mm TL were equivalent during both vegetated and nonvegetated periods. Young-of-year largemouth bass in both Lakes Pearl and Baldwin had first year growth averaging 150-mm TL during years when submersed vegetation was present. After macrophyte removal, first year growth averaged 210-mm TL in both lakes. Largemouth bass in Lake Pearl averaged 28 months to reach 250-mm TL during vegetated years. After all submersed vegetation was eliminated, largemouth bass grew to 250-mm TL in 18 months.

Discussion

Grass carp were successfully used for long-term elimination of both hydrilla and other submersed aquatic macrophytes in Lakes Baldwin and Pearl. Other urban Florida lakes (Lakes Bell (32 ha), Clear (64 ha), Holden (102 ha), Killarney (96 ha) and Wales (132 ha)) stocked with grass carp in the 1970s to control hydrilla have also been devoid of all submersed macrophytes since plant removal occurred (Leslie, Nall, and Van Dyke 1983; Leslie et al. 1987; and authors' personal observations). Plant species eliminated in the Florida lakes were bladderwort, common cattail, coontail, eel-grass, hydrilla, maidencane, pickerelweed, and torpedograss. Aquatic macrophytes still present in these systems include spatterdock, fragrant water lily, sawgrass, and elephant-ear (*Colocasia esculenta*). Complete removal of submersed macrophytes in these Florida lakes resulted in similar changes in water quality parameters that occurred in Lakes Baldwin and Pearl: increases in nutrient-related variables of total phosphorous, total nitrogen, chlorophyll *a*, and increased turbidity or a decrease in Secchi transparency (Leslie, Nall, and Van Dyke 1983; Leslie et al. 1987).

Complete removal of submersed plants from Lake Pearl caused significant reductions in total fish standing crop estimates and the elimination or reduction to levels below detection of six fish species. Lake Baldwin total standing crop estimates were not impacted by plant removal. Centrarchid populations, except for warmouth, were not adversely affected by the total removal of submersed macrophytes. Clupeid species in both lakes increased significantly with submersed plant removal. Utilization of grass carp for vegetation control in systems containing fish species that are protected or of special concern (Johnson 1987) or species known to require submersed macrophytes within their life cycles should be exercised with caution because long-term macrophyte removal may occur.

The actual duration of submersed vegetation control with grass carp has not been verified; fish are still present and controlling aquatic macrophytes in Florida's lakes stocked in the 1970s. Grass carp removal techniques by both State and Federal agencies and universities in Florida utilizing haul seines, gill nets, trammel nets, trap nets, electrofishing, and 0.1 rotenone treatments have not proven successful in reducing stocks to levels that resulted in submersed macrophyte regrowth in systems >20 ha. Therefore, utilization of grass carp for submersed vegetation control should be viewed as a long-term management decision; control will occur for a minimum of 15 years in lakes with surface water temperatures over 20 °C for at least 7 months of the year.

Table 6
Annual Mark-Recapture Estimates and 95-Percent Confidence Intervals for Florida Largemouth Bass in Lake Pearl, Florida

Total Length, mm	Year	Number/Hectare (kg/hectare)		
		Lower Bound	Estimate	Upper Bound
151 to 250	1980	14 (3.3)	26 (3.5)	52 (12.1)
	1981	41 (8.6)	86 (8.5)	199 (41.6)
	1982	39 (8.5)	96 (10.9)	241 (52.5)
	1983	48 (7.9)	69 (11.4)	103 (17.0)
	1984	13 (2.1)	21 (3.4)	36 (5.9)
	1985	10 (1.4)	22 (3.1)	56 (7.8
	1986	12 (1.6)	22 (3.0)	45 (6.0)
	1987	25 (1.7)	38 (5.4)	60 (8.5)
	1989	4 (0.5)	7 (0.8)	15 (1.7)
>250	1979	19 (8.9)	37 (17.4)	78 (36.7)
	1980	38 (15.9)	55 (23.1)	83 (34.8)
	1981	40 (23.7)	63 (37.4)	102 (60.5)
	1982	28 (25.3)	49 (44.40)	91 (82.4)
	1983	26 (18.5)	42 (30.0)	72 (51.3)
	1984	30 (12.2)	47 (19.1)	79 (32.1)
	1985	13 (8.8)	21 (14.2)	35 (23.7)
	1986	18 (14.3)	27 (21.5)	42 (33.4)
	1987	21 (13.1)	30 (18.8)	43 (26.9)
	1989	12 (7.2)	16 (9.6)	22 (13.1)
≥500	1979	0.5 (1.7)	1.3 (4.2)	3.3 (10.4)
	1980	0.8 (2.5)	2.3 (6.8)	5.7 (16.8)
	1981	1.5 (3.6)	5.0 (12.0)	9.1 (21.8)
	1982	1.1 (2.9)	2.4 (6.1)	5.4 (14.0)
	1983	0.5 (1.1)	1.1 (2.8)	2.7 (7.0)
	1984	0.3 (0.8)	0.8 (2.0)	2.0 (4.9)
	1985	0.3 (1.0)	0.9 (2.5)	2.1 (6.2)
	1986	1.0 (2.9)	3.4 (9.5)	6.1 (17.2)
	1987	0.2 (0.6)	0.6 (1.7)	1.5 (4.2)
	1989	0.3 (0.6)	0.8 (2.0)	1.5 (3.5)

Table 7
Mean Daily Growth Rates for Floy-Tagged Florida Largemouth Bass from Lake Pearl, Florida, 1979-1989

Total Length mm	Year	Sample Size		Growth	
		mm/day	gm/day	mm/day (mean ± 2SE)	gm/day (mean ± 2SE)
151 to 250	1979-1982	80	71	0.13 ± 0.02	0.24 ± 0.05
	1983-1986	177	169	0.22 ± 0.02	0.43 ± 0.04
	1987-1989	33	30	0.26 ± 0.08	0.60 ± 0.20
251 to 350	1979-1982	111	98	0.10 ± 0.02	0.40 ± 0.08
	1983-1986	210	194	0.16 ± 0.02	0.65 ± 0.10
	1987-1989	58	50	0.13 ± 0.02	0.59 ± 0.14
>350	1979-1982	37	31	0.10 ± 0.03	1.21 ± 0.56
	1983-1986	46	39	0.14 ± 0.04	1.48 ± 0.46
	1987-1989	33	29	0.08 ± 0.02	0.94 ± 0.24

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Effects of Grass Carp on the Aquatic Vegetation in Lake Conway, Florida

by
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The Lake Conway project was initiated in 1976 as a U.S. Army Corps of Engineers Large Scale Operations Management Test (LSOMT) of the feasibility of managing hydrilla (*Hydrilla verticillata*) with grass carp (*Ctenopharyngodon idella*) and to delineate possible collateral effects on aquatic ecology (see Addor and Theriot 1977). The Florida Department of Environmental Protection (FDEP), formerly the Department of Natural Resources, was contracted to evaluate the effects of grass carp on hydrilla and nontarget aquatic vegetation.

Lake Conway is a 737-ha lake located in Orlando, FL (Nall and Schardt 1977). The lake is composed of five interconnected pools consisting of Lake Gatlin, Little Lake Conway (east and west pools) and Lake Conway (middle and south pools). Lake Conway is an urban representative of the Central Ridge waters of Florida. Canfield (1981) reported the following water quality values: pH 7.4 to 7.7; total alkalinity 27 to 31 mg/L $CaCO_3$; total nitrogen 350 to 517 µg/L; total phosphorus 3.7 to 24.2 µg/L; chlorophyll *a* 1.7 to 6.7 µg/L; color 0 to 5 mg/L as Pt; Secchi transparency 3.4 to 4.0 m. The typical shoreline plant species of cattail (*Typha latifolia*), maidencane (*Panicum hemitomon*), and torpedograss (*P. repens*) had been removed around much of the lake by homeowners at the inception of this study (Nall and Schardt 1977). Illinois pondweed (*Potamogeton illinoensis*), nitella (*Nitella megacarpa*), and tapegrass (*Vallisneria americana*) were the dominant submersed species; however, by 1976, hydrilla was rapidly expanding.

Methods

The Lake Conway system was stocked in September 1977 with an average of 10.4 monosex (all female, Stanley 1976) grass carp per hectare (Table 1). This stocking rate represents approximately 20 fish per hectare of submersed vegetation and 1.6 fish per metric ton of vegetation present in the lake at the time of stocking (Table 1). Stocking rate figures for each pool were determined by an experimental stocking rate model that considers vegetation biomass, grass carp feeding rates, and other parameters in its computations (Nall and Schardt 1977). Because the pools are interconnected, grass carp can move freely among them. Such movement is much more restricted between the middle pool and both south and east pools than between the east and west pools, which share a 4-m-deep, 50-m-wide connection (Nall et al. 1979).

Random plant biomass sampling

Samples were collected using a 7.7-m length pontoon barge, with a hydraulic sampling device that was specifically designed for this project. A map of the four major pools (Lake Gatlin was excluded for logistical reasons) was overlaid with a numbered grid. Sixty random samples were chosen for each pool by selecting numbered squares designated by random number tables. The 0.25-m^2 sampling device was lowered and retrieved hydraulically. Each sample was separated by species and washed to remove mud and detritus. Excess water was removed by vigorous shaking. Fresh weight was determined to the nearest gram.

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Table 1
Stocking Rates of Monosex Diploid Grass Carp Placed into Lake Conway, Florida, on September 9, 1977 (Also included are data on stocking of triploid grass carp in 1986 and 1988)

Stocking	South	Middle	East	West	Overall
September 9, 1977					
Total grass carp	1,629	3,408	919	1,066	7,022
Grass carp/ha	12.3	12.3	7.4	7.4	10.4
Grass carp/ha veg	17.8	24.2	16.0	16.0	19.9
Grass carp/tonne veg	2.1	1.7	1.2	1.2	1.6
			East-West		
1986					
Total grass carp	—	—	1,600		1,600
Grass carp/ha	—	—	6.0		2.4
Grass carp/ha veg	—	—	9.9		3.7
Grass carp/tonne veg	—	—	1.7		0.8
1988					
Total grass carp	—	—	1,000		1,000
Grass carp/ha	—	—	3.7		1.5
Grass carp/ha veg	—	—	4.3		2.0
Grass carp/tonne veg	—	—	0.4		0.2

SCUBA plots

Fifteen permanently marked 0.1-ha plots were located in selected areas of the four major pools of Lake Conway. A SCUBA diver swam in a random underwater pattern in each plot while making 30 visual observations. Percent frequency of occurrence for each species was calculated from these data.

Results and Discussion

Baseline sampling showed that submersed plants were rarely encountered at depths below 6 m. Approximately 35 percent of south, west, and middle pools and 17 percent of east pool are deeper than 6 m (Nall and Schardt 1977).

A summary of average percent frequency of occurrence of hydrilla versus all other aquatic plant species from SCUBA plots monitored from July 1976 to August 1980, then bimonthly to August 1981, is presented in Figure 1. Also included in Figure 1 is the total number of species encountered. These data show that the impact of the grass carp was relatively slow in this study—no overt changes were noted until the second summer after stocking. Two years after stocking 10.4 grass carp/ha, hydrilla was greatly reduced with no discernable effect on frequency of occurrence of other species. Total number of species was slightly higher during the poststocking period.

Hydrilla, Illinois pondweed, and nitella are highly preferred foods for grass carp, but tapegrass is nonpreferred (Nall and Schardt 1977; Leslie et al. 1987). Data from SCUBA plots showed that tapegrass increased dramatically as the pressure of herbivory reduced the competitiveness of the other species (Figure 2). The expansion of tapegrass was considered desirable because it is considered to be an ecologically valuable native plant (Duke and Chabreck 1976). If instead this had been an unpalatable, canopy-forming, nonnative species, such as Eurasian watermilfoil (*Myriophyllum spicatum*), such expansion would be highly undesirable, altering the fishery and the traditional uses of this water body (Van Dyke, Leslie, and Nall 1984; Colle et al. 1987; Leslie et al. 1987). During this time period, nitella remained relatively stable. Illinois pondweed increased initially until hydrilla became scarce, then the grass carp began to feed on Illinois pondweed and reduced its abundance.

SCUBA monitoring resumed in 1984 with FDEP funds upon completion of the U.S. Army Corps of Engineers contract period.

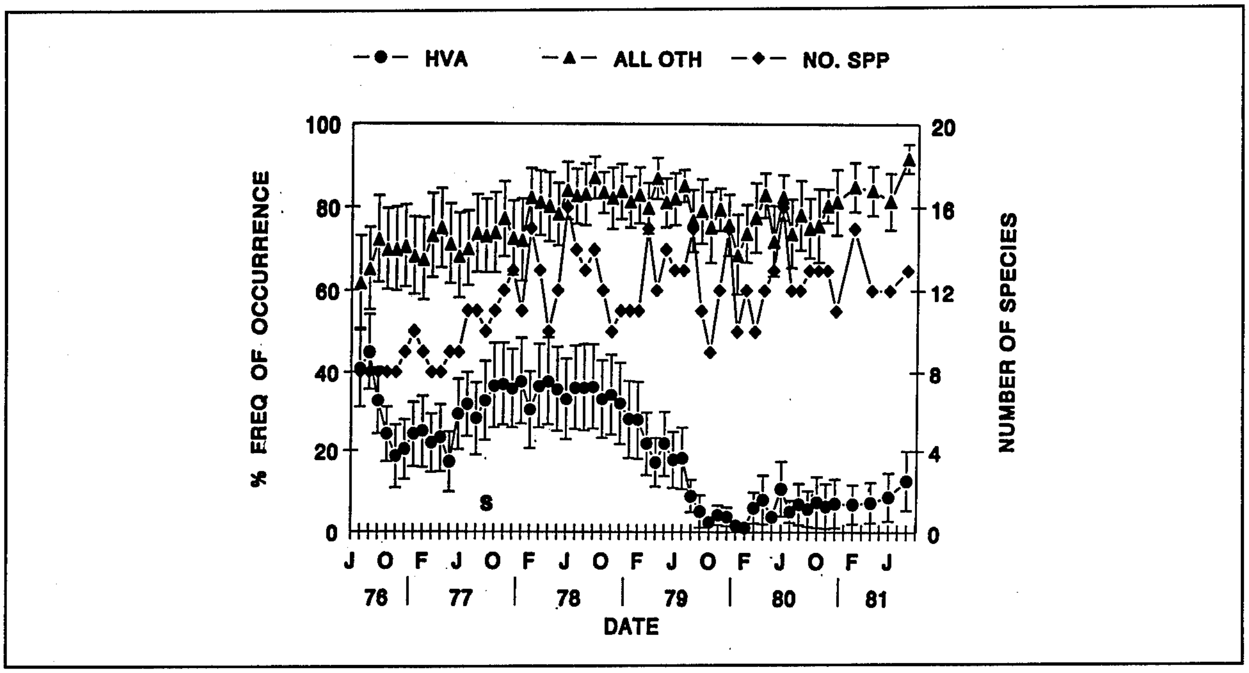

Figure 1. Average percent frequency of occurrence of hydrilla (HVA) and all-other-submersed aquatic plants in fifteen 0.1-ha SCUBA plots located throughout the Lake Conway chain (east, west, middle, and south pools), Florida. Thirty random observations were made per plot each month (bimonthly in 1981). NO. SPP = total number of species observed each sampling date. Error bars represent one standard error of the mean 1977. Monosex (female) grass carp (10.4)/ha were stocked in September 1977

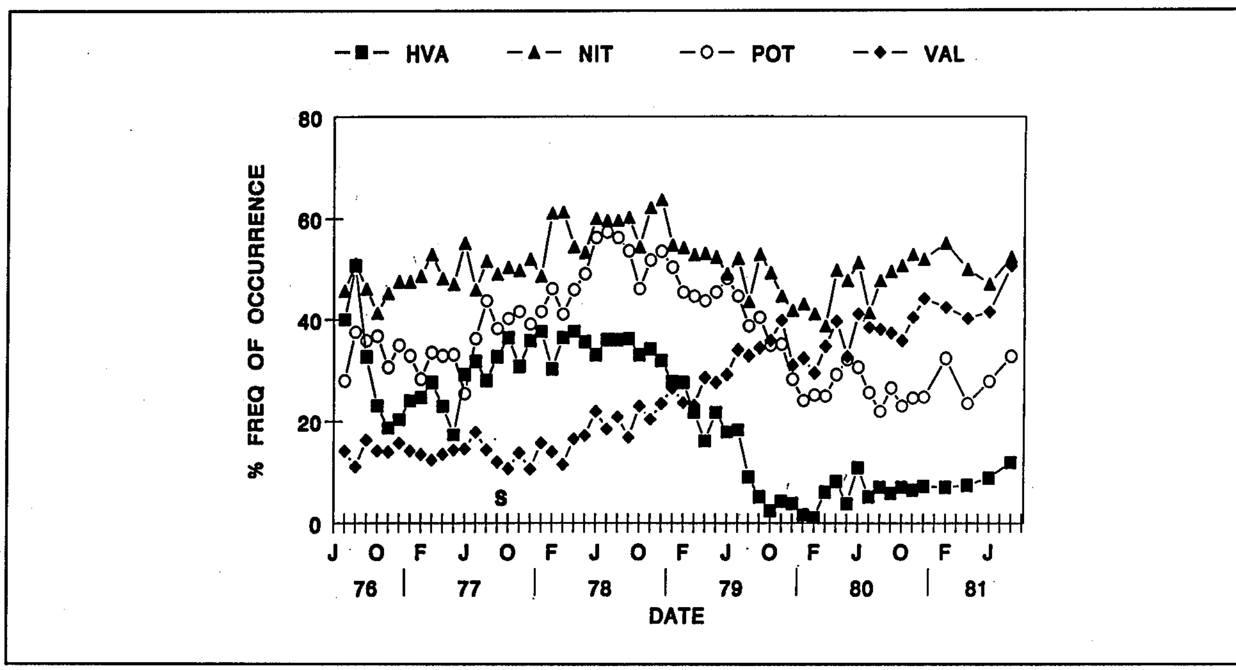

Figure 2. Average percent frequency of occurrence of the four predominant submersed aquatic plants in fifteen 0.1-ha SCUBA plots located throughout the Lake Conway chain (east, west, middle, and south pools), Florida. Thirty random observations were made per plot each month (bimonthly in 1981). HVA = hydrilla, NIT = nitella, POT = Illinois pondweed, VAL = tapegrass. Monosex (female) grass carp (10.4)/ha were stocked in September 1977

SCUBA plots have since been monitored annually in late summer. An additional 2.4 and 1.5 triploid grass carp/ha were stocked in 1986 and 1988, respectively, in response to the increased frequency of occurrence of hydrilla (Figure 3). These stockings reduced hydrilla by 1989, and hydrilla frequency of occurrence has remained low to date. There has been little discernable effect on frequency of occurrence of other submersed species; but a drop in the number of species was detected.

The dynamics of the four major species on SCUBA plots over the last 17 years are shown in Figure 4. The percent frequency of occurrence of tapegrass increased from about 10 percent in 1979 to an average of more than 50 percent over the last 10 years. Illinois pondweed and nitella, both highly palatable species, have remained fairly abundant throughout the study. Filamentous algae are increasing in the Lake Conway system and are indicative of increased urbanization in the watershed.

Figure 5 depicts the results of random plant biomass sampling conducted in late summer each year. Note that in Lake Conway, hydrilla was not a dominant component of the aquatic plant community at any time during the study. As seen with the SCUBA data, hydrilla biomass was reduced by more than 99 percent in 2 years without reducing the overall biomass of other species. By 1984, hydrilla biomass began to rebound, but was again reduced by the stocking of additional triploid grass carp in 1986 and 1988. These stockings resulted in no apparent reduction in overall biomass of other submersed species. Percent frequency of occurrence of plants other than hydrilla in the biomass samples (Figure 6) shows an increasing trend not seen in the SCUBA data (Figure 1).

An examination of the biomass data for the four major plant species in Lake Conway over time reveals that tapegrass has increased, nitella has declined slightly, and that Illinois pondweed (a preferred food species for grass carp) showed initial declines, but has increased

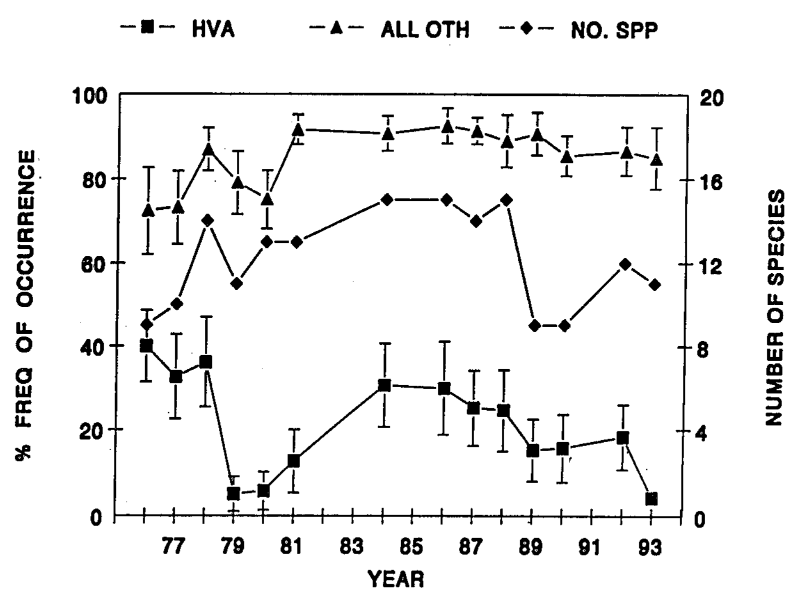

Figure 3. Average percent frequency of occurrence of hydrilla (HVA) and all-other-submersed plants in fifteen 0.1-ha SCUBA plots located throughout the Lake Conway chain (east, west, middle, and south pools), Florida. Thirty random observations were made in each plot in late summer each year. NO. SPP = total number of species encountered per sampling date. Error bars represent one standard error of the mean. Monosex (female) grass carp (10.4)/ha were stocked in September 1977. In December 1986 and 1988, 6.0 and 3.7 triploid grass carp/ha, respectively, were stocked into east-west pool

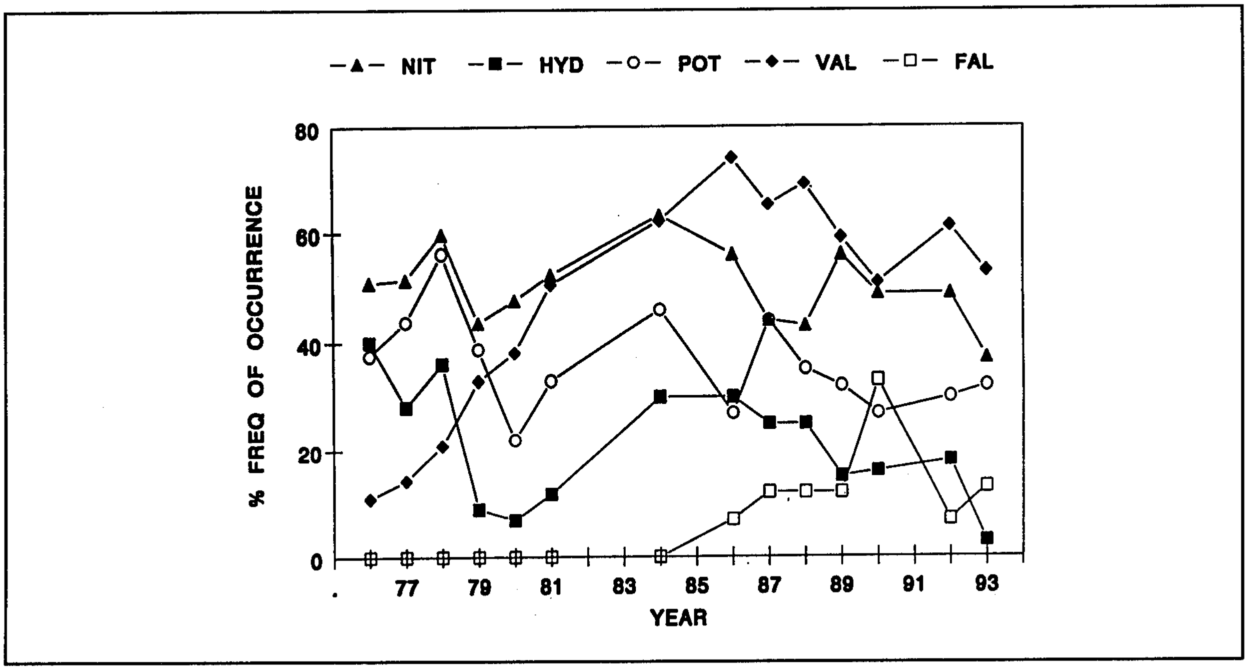

Figure 4. Average percent frequency of occurrence of five submersed aquatic plants in fifteen 0.1-ha SCUBA plots located throughout the Lake Conway chain (east, west, middle, and south pools), Florida. Thirty random observations were made per plot in late summer each year. NIT = nitella, HYD = hydrilla, POT = Illinois pondweed, VAL = tapegrass, FAL = filamentous algae. Monosex (female) grass carp (10.4)/ha were stocked in September 1977. In December 1986 and 1988, 6.0 and 3.7 triploid grass carp/ha, respectively, were stocked into east-west pool

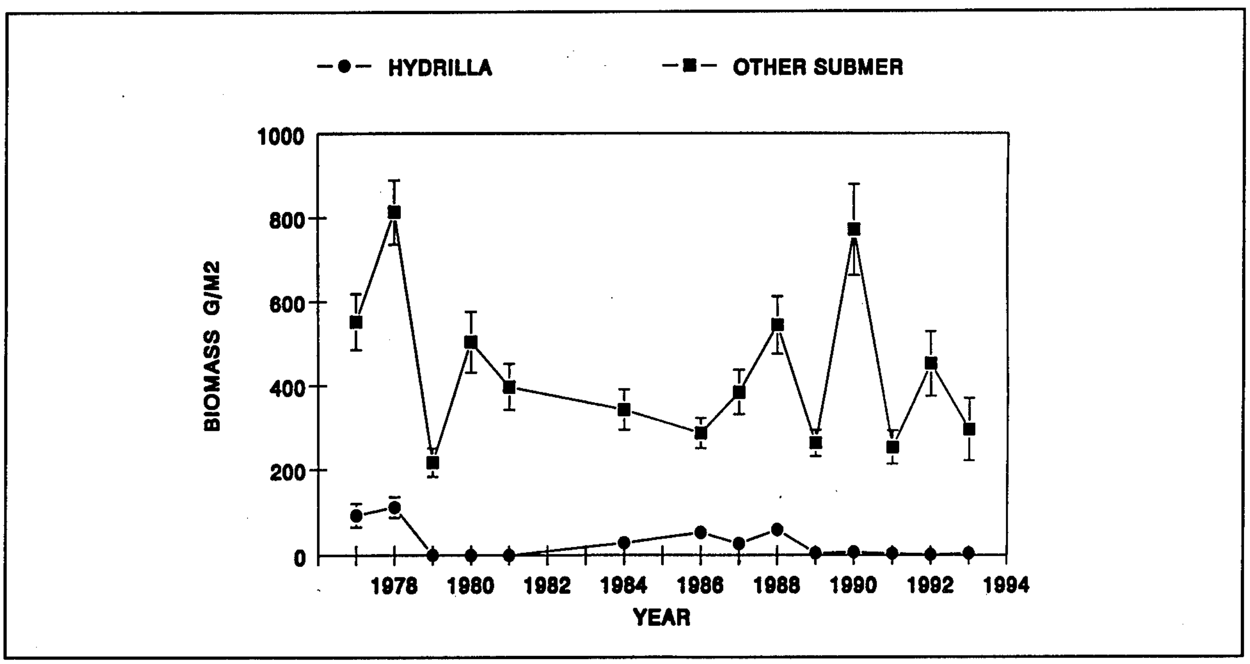

Figure 5. Fresh weight biomass (g/m²) of hydrilla and all-other-submersed aquatic plants collected in random biomass samples in the Lake Conway chain (east, west, middle, and south pools), Florida. Sixty random samples were collected in each pool in late summer each year. Error bars represent one standard error of the mean. Monosex (female) grass carp (10.4)/ha were stocked in September 1977. In December 1986 and 1988, 6.0 and 3.7 triploid grass carp/ha, respectively, were stocked into east-west pool

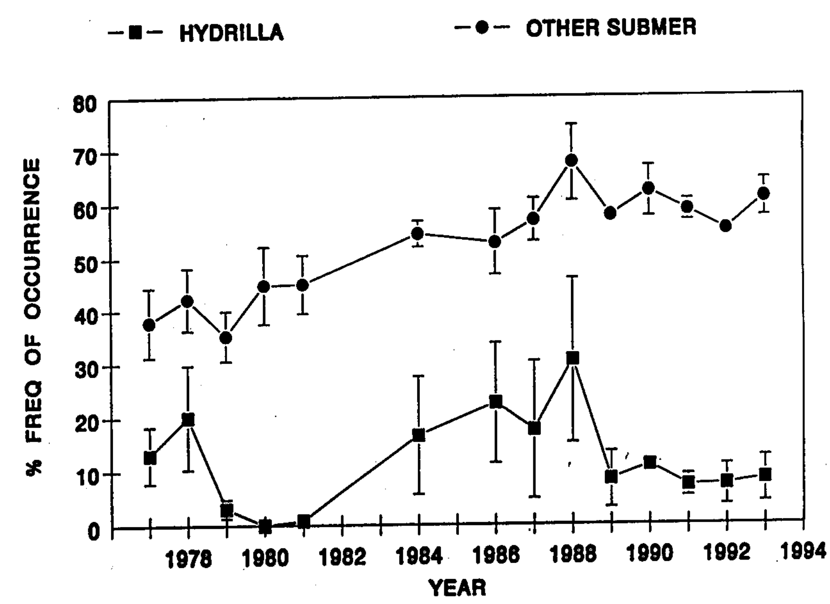

Figure 6. Percent frequency of occurrence of hydrilla and all-other-submersed aquatic plants collected in random biomass samples in the Lake Conway chain (east, west, middle, and south pools), Florida. Sixty random samples were collected in each pool in late summer each year. Error bars represent one standard error of the mean. Monosex (female) grass carp (10.4)/ha were stocked in September 1977. In December 1986 and 1988, 6.0 and 3.7 triploid grass carp/ha, respectively, were stocked into east-west pool

over the latter years of the study (Figure 7). Grass carp have been able to maintain hydrilla at low biomass for over 15 years with minimal impact on other submersed species.

In many instances, plant management with grass carp has resulted in total (or near-total) elimination of vegetation or has had little impact at all. This all-or-none response has not been the case in Lake Conway. But we have not been able to duplicate these results in other lakes over the long term. Reasons are elusive but some factors to consider include the following: (a) although perhaps locally abundant, hydrilla was a relatively minor component of the submersed plant community in Lake Conway at the onset of this LSOMT; (b) the lake, which was treated with endothall herbicide 1 year prior to the onset of this project, historically had an abundant and diverse submersed native plant community; (c) there were at least three abundant highly palatable plant species present, including hydrilla; (d) there were less palatable native species, like tapegrass, poised to fill vacant space; and (e) the lake is relatively large, clear, and deep.

Figure 8 depicts three central Florida lakes that, while submersed plants dominated, were so because of a dense, near-monoculture of hydrilla (Van Dyke, Leslie, and Nall 1984). Under these conditions, perhaps any stocking rate that affected hydrilla or even use of integrated methods with low stocking rates would result in lakes devoid of submersed vegetation. In Lake Conway, however, grass carp have maintained hydrilla at low levels, while an abundant and diverse native submerged flora has continued to flourish.

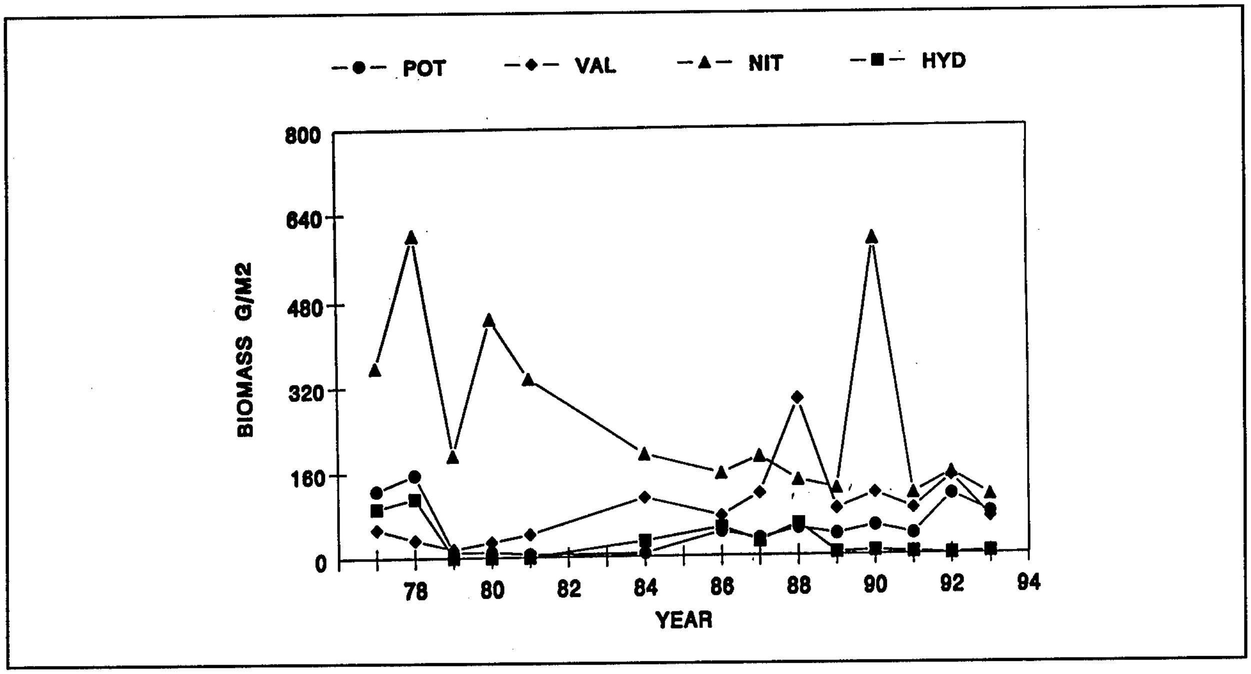

Figure 7. Fresh weight biomass (g/m^2) of the four predominant submersed aquatic plants collected in the Lake Conway chain (east, west, middle, and south pools), Florida. Sixty random samples were collected in each pool in late summer each year. POT = Illinois pondweed, VAL = tapegrass, NIT = nitella, HYD = hydrilla. Monosex (female) grass carp (10.4)/ha were stocked in September 1977. In December 1986 and 1988, 6.0 and 3.7 triploid grass carp/ha, respectively, were stocked into east-west pool

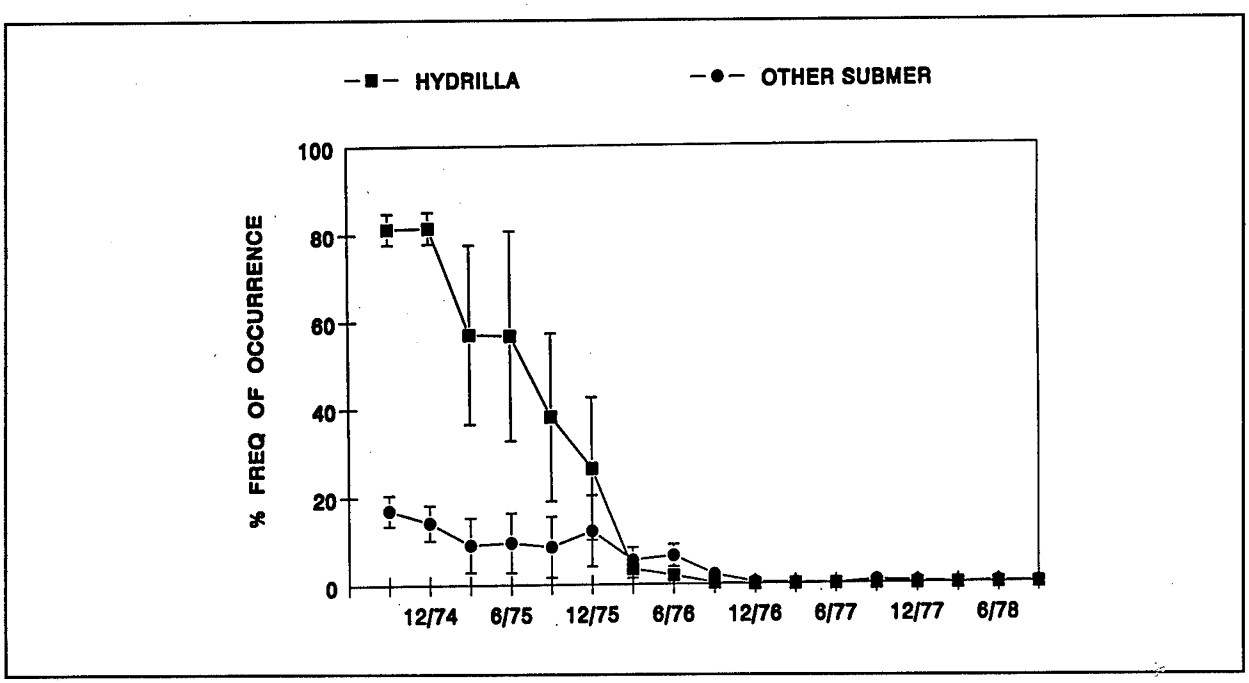

Figure 8. Average percent frequency of occurrence of hydrilla and all-other-submersed aquatic plant species on transects monitored quarterly in three central Florida lakes. Lakes Bell, Clear, and Holden were stocked with 50 diploid grass carp/ha in October 1974. Error bars represent one standard error of the mean

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Effects of Triploid Grass Carp and Sonar Treatments on Aquatic Plants in Lake Yale

by

Rue S. Hestand III,[1] Boyd Z. Thompson,[1] and Craig T. Mallison[1]

Introduction

Hydrilla *(Hydrilla verticillata)* has become a problem in many large Florida lake systems, and traditional control with herbicides is very costly. Triploid grass carp *(Ctenopharyngodon idella)* are efficient, functionally sterile herbivores. Information on using this fish to control hydrilla without detrimentally affecting desirable plants in large water bodies of Florida is generally lacking. Lake Conway, a 430-ha lake located in Orange County, Florida, is the only large lake where diploid grass carp have been studied with the objective of leaving desirable submerged aquatic plants while controlling hydrilla (Nall and Schardt 1978). This lake was similar to Lake Yale in that it had a large native aquatic plant population dominated by Illinois pondweed *(Potamogeton illinoensis)*. Lake Conway was stocked in September 1977 with supplemental stockings in 1984 and 1987. To date, the grass carp have successfully controlled hydrilla but temporarily eliminated Illinois pondweed. Submerged aquatic plants remain in the lake, but the population makeup has been altered.

The objective of this study was to determine if triploid grass carp stocked at seven fish/hectare of lake area could control[2] hydrilla regrowth without detrimentally impacting the submerged native plants.

Study Area

Lake Yale is a 1,636-ha lake located in Lake County near Eustis, FL. Hydrilla became established in the southern end of Lake Yale in 1980. The dominant submerged plant at that time was Illinois pondweed, which covered approximately 140 ha and ranged to a depth of 3 m. In 1981, 22 ha of hydrilla were treated with Diquat plus copper and Aquathol K. In 1983, the hydrilla had expanded to 1,200 ha, and 252 ha were treated with various herbicides including Sonar (fluridone). These treatments destroyed the spatterdock (*Nuphar luteum)* that had dominated the south end of the lake. In 1984, 1,400 ha of hydrilla were reported, and a large-scale Sonar treatment was scheduled. Ten 10-ha plots were treated with Sonar (310 kg active) in April 1984. By October 1984, only 128 ha of hydrilla remained, and pondweed had expanded to 320 ha. No hydrilla was found after the October 1984 survey until March 1987, when approximately 2 ha were found in the south end of the lake. At that time, pondweed had expanded to 1,080 ha and reached a depth of 5 m. Between July 1987 and February 1989, 14,945 triploid grass carp were stocked (Table 1). Hydrilla expansion continued; in January 1990, 42 ha of dense hydrilla in the south end of the lake were treated with Aquathol K, and 6,200 additional triploid grass carp were stocked. The hydrilla continued to expand until treated with Sonar in 1992 and 1993. These treatments virtually eliminated hydrilla, and the grass carp have severely impacted the remaining submerged plant population.

Materials and Methods

Six permanent transects were established in 1987 to representatively sample plant communities in Lake Yale (Figure 1). Point samples were taken at 15-m intervals along each

[1] Florida Game and Freshwater Fish Commission, Eustis, FL.

[2] Target plant is kept below problematic levels so that it does not interfere with the intended use(s) of that water body.

Table 1
Date and Number of Triploid Grass Carp Stocked in Lake Yale, Hydrilla Coverage, and Frequency of Occurrence (Total fish equals fish stocked minus estimated mortality)

Date	Fish Stocked	Total Fish	Hydrilla Coverage, ha	Hydrilla Frequency of Occurrence, %
3/87			2	
5/87			41	
7/87	5,400	1,200	81	20
11/87	750	1,950	90	21
12/87	775	2,537	90	
1/88	1,000	3,386	100	
2/88	550	3,777		26
5/88	1,500	5,262	130	
6/88	2,100	6,982		30
10/88	1,400	7,824	162	25
2/89	1,470	8,885	162	28
5/89		8,489	196	29
8/89		8,332	410	30
11/89		8,272	520	32
1/90	6,200	13,433	800	53
12/90		12,188	1,310	81
12/91		11,304	1,340	83
12/92		10,394	650	63
12/93		9,237	0	0
Total	**21,145** **13/hectare**	**—** **6/hectare**		

line quarterly. Plants were sampled using a pole with hooks that was lowered through the water and rotated. Percent frequency of occurrence was recorded of all plants collected. A recording fathometer (Raytheon DE-719B) was also run quarterly on eight additional transect lines (totaling 17 km) to estimate total coverage of aquatic plants (Figure 1). Biomass sampling (Nall and Schardt 1978) was conducted semiannually starting in 1989 to determine biomass (metric tonnes) of each plant species. Vegetation maps for major plant species were constructed using the Loran coordinates of 200 biomass sampling stations.

Mortality estimates from the first triploid grass carp stocking in July 1987 were made by stocking a subsample of fish in ponds at the Richloam Hatchery. Thereafter, estimates of the number of fish remaining in the lake were corrected using an estimate of 25-percent mortality for the first year after stocking, 5 percent each of the next 2 years, 10 percent each year of the next 3 years, and 25 percent per year thereafter (Hestand, Thompson, and Clapp 1990).

In 1992, 200 kg (active ingredient) of Sonar was applied to the northwest portion of Lake Yale with biweekly treatments from January 13 to March 30 (Figure 1). Water samples to be analyzed for fluridone were collected biweekly through June and once in July, September, and October by Lake County Water Authority personnel. These samples were analyzed for fluridone by Dr. John Rodgers, University of Mississippi, Biological Field Station.

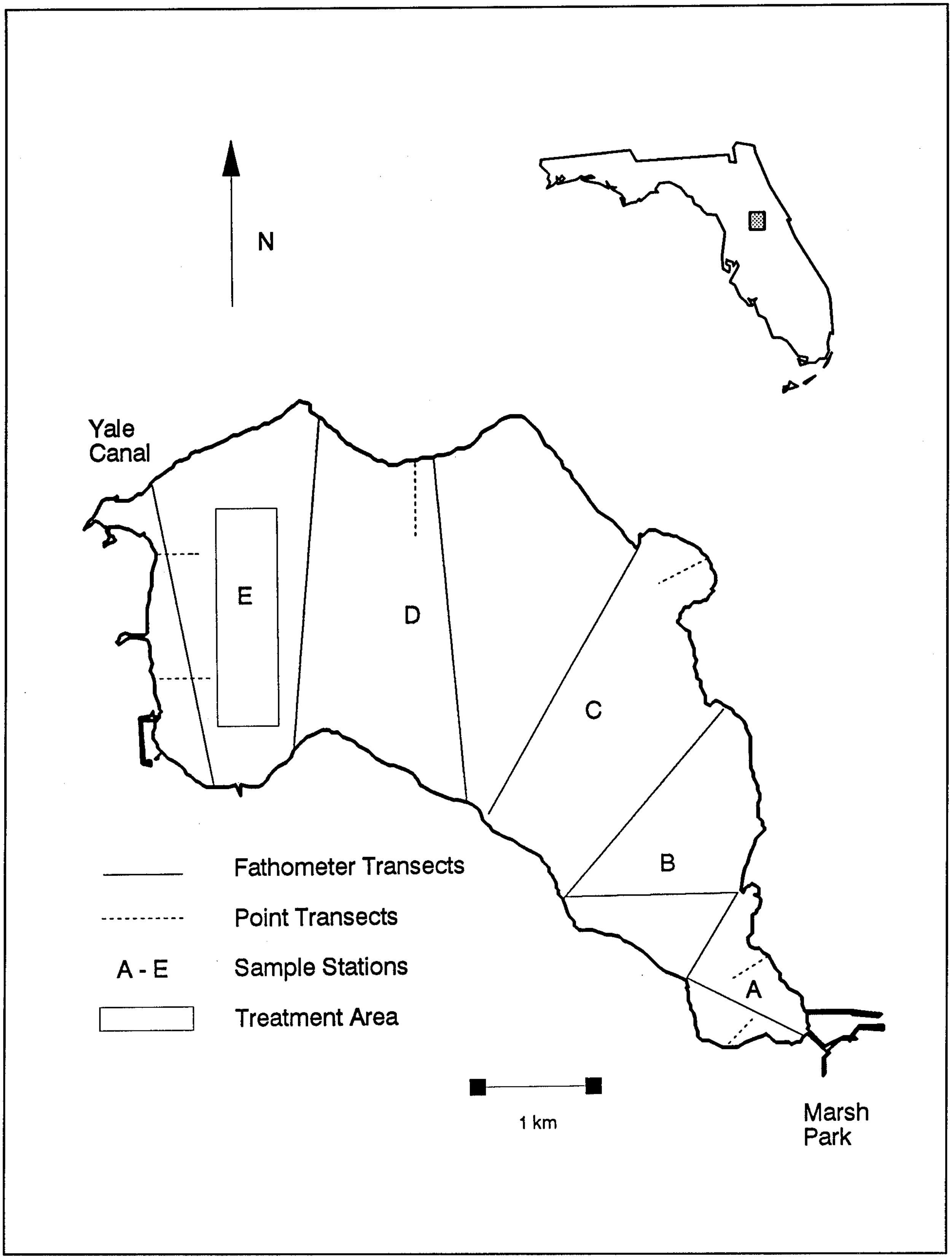

Figure 1. Map of Lake Yale illustrating the location of recording fathometer point transects, fluridone sample stations, and treatment area

In 1993, 181 kg (active ingredient) of Sonar was placed in the same area of Lake Yale as the 1992 treatment. However, this time the treatment was split into three applications over a 2-week period covering February and March.

Results and Discussion

The hydrilla that covered 2 ha in March 1987 had expanded to 41 ha by May 1987. The stocking of 3.7 triploid grass carp/hectare (146 fish/hectare of hydrilla) originally planned for May 1987 was delayed until mid-July 1987. Of the 6,000 fish ordered, 600 were placed in three ponds at the Richloam Hatchery (Florida Game and Fresh Water Fish Commission hatchery near Webster), where mortality ranged from 74 to 82 percent within 30 days. This mortality was likely due to shipping stress and stocking into warm water (31 °C). Thus, of the 5,400 fish stocked in July 1987, we estimated that only 1,200 fish survived longer than 2 weeks (15 fish/hectare of hydrilla). An additional 1,525, 6,550, 1,470, and 6,200 triploid grass carp were stocked in 1987, 1988, 1989, and 1990, respectively. Thus the highest density of triploid grass carp during the course of this study (excluding mortality) was 13/lake hectare or 20/hydrilla hectare in January 1990. Factoring in estimated mortality reduced these densities to eight/lake hectare or 17/hydrilla hectare in January 1990 (Table 1). Earlier work with diploid grass carp indicated that a minimum of 123 fish per hectare of hydrilla or 20 per metric ton of plants were necessary to eliminate hydrilla (Osborne 1982).

Hydrilla was concentrated in the south end of the lake (Marsh Park, Figure 1) during 1987 (90 ha). It expanded each year until December 1989 when it occurred in approximately 500 ha. During this same time period, there was a slow decline in coverage of Illinois pondweed that probably resulted from two factors: competition from expanding hydrilla and grazing from the triploid grass carp. On one transect, Illinois pondweed had a frequency of occurrence in November 1987 of 75 percent, but by November 1989 had declined to 10 percent, while hydrilla frequency of occurrence increased to 95 percent. The ability of hydrilla to exclude native plants has been described by Bowes et al. (1977), Haller and Sutton (1975), Van et al. (1977), Van, Haller, and Bowes (1976), and Van, Haller, and Garrard (1978). Even though triploid grass carp prefer hydrilla and feed heavily on it, they also consume Illinois pondweed (Figure 2). Hydrilla had a frequency of occurrence of 30 percent in the lake, but it was found in 90 percent of the triploid grass carp stomachs that were analyzed. Frequency of occurrence of Illinois pondweed was 42 percent in the lake, and it was found in 45 percent of the stomachs. Stomach analysis may be biased, as triploid grass carp were more easily collected where hydrilla was abundant. Telemetry data probably yielded a better picture of what the fish were eating. Hydrilla was found at 61 percent of radio telemetric locations of triploid grass carp and pondweed at 49 percent of the locations (Hestand, Thompson, and Clapp 1989).

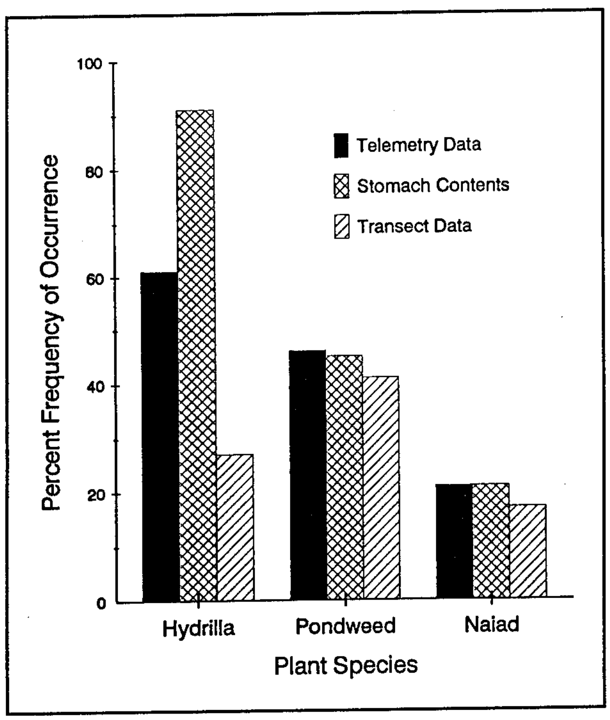

Figure 2. Estimated plant consumption by triploid grass carp based on stomach contents and carp location (telemetry) data. Transect data estimates relative abundance of major plant types

By October 1991, hydrilla had increased to 83-percent frequency of occurrence in biomass samples and made up 74 percent of the total plant biomass. The biomass and percent frequency of occurrence of Illinois pondweed showed little change from the severe decline in 1990. The biomass of southern naiad (*Najas quadalupensis*) was reduced by 50 percent, while the frequency of occurrence declined by 20 percent from November 1990, probably from competition from hydrilla. Southern naiad biomass increased each winter, then declined with the increase in hydrilla during the summer. The reduction in biomass through 1994 was probably due to grass carp feeding (Table 2).

Table 2
A Comparison of Biomass (MT = metric ton) and Percent Frequency of Occurrence (Freq) of Dominant Vegetation in Lake Yale

Date	Hydrilla		Illinois Pondweed		Southern Naiad		Total Plants	
	MT	Freq	MT	Freq	MT	Freq	MT	Freq
Apr 93	996	12	3,388	52	465	16	5,646	65
Oct 89	1,063	25	5,978	68	2,524	14	11,491	81
Apr 90	863	32	465	42	1,926	35	4,849	77
Oct 90	2,160	53	262	24	3,337	51	6,871	82
Mar 91	3,361	61	133	17	3,720	69	7,705	88
Oct 91	11,624	83	300	20	1,706	41	15,804	90
Apr 92	10, 916	82	280	13	2,106	43	15,618	93
Sep 92	2,143	63	1,014	22	1,276	45	7,280	89
Apr 93	355	40	170	17	2,215	55	3,988	71
Oct 93	1	2	175	9	1,789	30	3,569	48
Apr 94	2	2	1	0.4	2,377	24	4,149	36
Oct 94	3	2	0.3	0.4	723	19	3,405	40

While the coverage of hydrilla continued to expand, Illinois pondweed suffered a dramatic decline in coverage and biomass. Following Sonar treatments in 1992 and 1993, hydrilla was virtually eliminated. The 1992 treatment resulted in an 80-percent reduction in hydrilla biomass by September 1992 (Table 2). This lakewide reduction was not expected since the calculated theoretical concentration of fluridone was 3.4 ppb, and a minimum of 5 ppb was thought to be needed to control hydrilla. Current work has found that mature hydrilla required exposure in excess of 500 FEDs (fluridone exposure days) to control hydrilla with significant regrowth appearing within 6 months (Fox, Haller, and Shilling 1994). Over 700 FEDs were required to repress regrowth for over 1 year. With FEDs of 350, only a 6-month suppression of hydrilla occurred. FEDs on Lake Yale ranged from 83 to 439 (Figure 3). However, when the standard deviation is figured, the high end of the confidence interval was 361 to 717 FEDs. This is within the necessary range for control and resulted in an 80-percent reduction in hydrilla biomass. Because of extensive regrowth, another treatment was scheduled in 1993. This Sonar treatment removed all hydrilla regrowth resulting in the grass carp reducing Illinois pondweed even more. In Lake Conway, Illinois pondweed virtually disappeared after hydrilla was eliminated (Schardt and Nall 1981). Stonewort (*Nitella* sp.) became the dominant plant following removal of hydrilla and Illinois pondweed by triploid grass carp. Subsequently, Illinois pondweed did return, but never reached its former density. In Lake Yale, southern naiad (*Najas guadalupensis*) and muskgrass (*Chara* sp.) are replacing Illinois pondweed (Table 3). However, their

Table 3
Respective Percent Frequency of Occurrence of Major Plant Species Sampled on Transect Lines in Lake Yale Between November 1987 and February 1994

Date	Illinois Pondweed	Coontail	Hydrilla	Southern naiad	Eelgrass	Bladderwort	Muskgrass	Bare
11/87	46.7	4.2	21.1	11.1	0.3	4.3	15.1	26.1
3/88	48.6	4.2	25.5	15.2	0.8	5.0	9.8	26.8
11/88	42.3	4.9	25.4	16.7	0.8	6.2	18.5	16.4
2/89	38.9	6.8	27.8	17.6	1.2	8.3	15.1	19.3
11/89	37.1	5.6	31.8	15.6	1.7	7.8	16.3	13.3
2/90	30.1	4.2	24.8	19.4	0.5	7.9	22.0	15.8
11/90	17.2	2.3	39.1	43.2	2.8	11.2	19.5	11.2
2/91	18.2	2.7	38.9	43.2	2.9	9.7	18.5	17.0
11/91	18.2	1.7	62.8	27.9	2.5	10.1	15.9	22.3
2/92	19.3	4.1	50.1	26.2	1.2	6.9	12.2	21.2
11/92	18.9	5.3	31.7	30.0	2.1	13.8	14.8	26.6
2/93	15.5	6.4	21.2	34.3	1.4	12.6	14.9	37.1
11/93	0.6	3.3	0	24.2	2.3	8.7	14.8	54.0
2/94	0.5	2.9	0.7	25.0	4.4	3.9	14.7	53.9
11/94	0	2.9	0.3	14.0	4.8	0.8	14.7	64.7

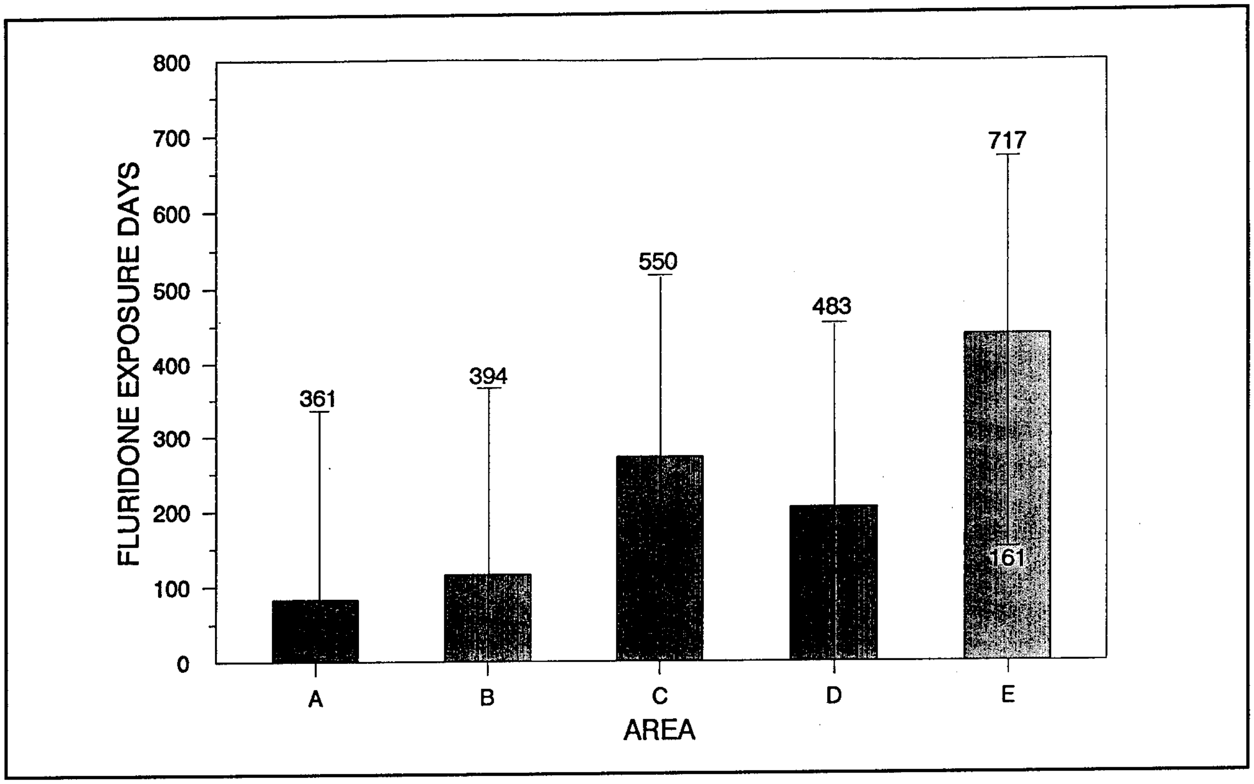

Figure 3. Fluridone exposure days at different sites in Lake Yale following a 1992 Sonar treatment

respective biomass will not sustain the number of grass carp in the lake. There are approximately five grass carp per metric ton (MT) preferred food and 2.6/MT of total submerged plants (Figure 4). If mortality is excluded, there are six/MT, which is enough grass carp to decimate the aquatic plant population (Leslie et al. 1987).

Conclusions

Because of the continued expansion of hydrilla, a stocking rate of 13 triploid grass carp/hectare (8/hectare adjusted for mortality) or 17/hydrilla hectare (adjusted) did not prevent the regrowth of hydrilla to problematic levels in Lake Yale. The Sonar treatments in 1992 and 1993 eliminated hydrilla, putting the native submerged plants at risk of being totally consumed by the triploid grass carp. It is possible that given the variability in the growth rate of hydrilla, it may be impossible to fine tune the triploid

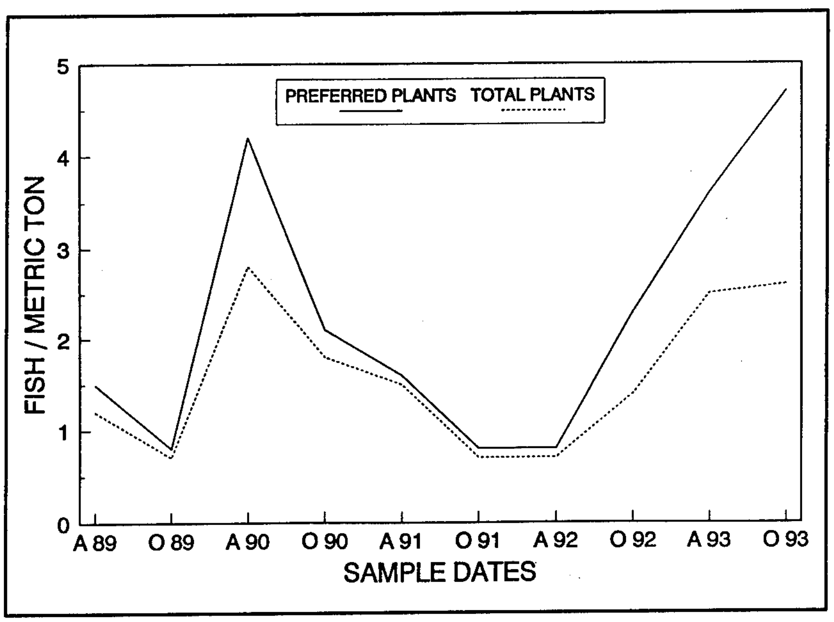

Figure 4. Comparison of number of triploid grass carp per metric ton of submerged plants to preferred plants in Lake Yale

grass carp stocking rate and not negatively impact preferred native plants that occur in lakes with hydrilla regrowing over the entire lake.

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Response of Fish Populations to Aquatic Plant Management in Lake Yale

by

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Introduction

The importance of aquatic plants to fisheries has been studied extensively with variable results (Hoyer et al. 1985; Hinkle 1986). Many studies have indicated that moderate levels (10 to 40 percent) of aquatic vegetation may be beneficial to fisheries (Colle and Shireman 1980; Durocher, Provine, and Kraai 1984; Wiley et al. 1984; Hinkle 1986). Some have indicated that removal of macrophytes can be detrimental to game fish communities (Ware and Gasaway 1976; Durocher, Provine, and Kraai 1984; Porak et al. 1990), while inconsistent results have led others to believe removal of aquatic plants may not adversely affect fisheries (Moorman 1956; Bailey 1978; Stanley, Miley, and Sutton 1978; Hoyer et al. 1985).

Effects of dense vegetation on largemouth bass *(Micropterus salmoides)* may include reduced foraging efficiency (Savino and Stein 1982; Colle et al. 1987), lower standing crop (Borawa et al. 1978; Wiley et al. 1984), reduced condition (Colle and Shireman 1980), or no change in the harvestable population (Colle et al. 1987). Food availability and condition of sunfishes was lower in dense aquatic vegetation (Bailey 1978; Borawa et al. 1978; Colle and Shireman 1980). Too little vegetation may also result in a lower standing crop of largemouth bass (Ware and Gasaway 1976; Smith and Crumpton 1977; Moxley and Langford 1982; Durocher, Provine, and Kraai 1984; Wiley et al. 1984; Porak et al. 1990). High levels of aquatic macrophytes generally resulted in higher recruitment of largemouth bass (Ware and Gasaway 1976; Moxley and Langford 1982; Strange, Kittrell, and Broadbent 1982; Klussman et al. 1988; Porak et al. 1990). However, constant or improved recruitment has also been observed after aquatic macrophyte removal (Hoyer et al. 1985). Forage fish populations generally increased at greater coverages of aquatic plants (Borawa et al. 1978; Moxley and Langford 1982; Wiley et al. 1984; Klussman et al. 1988). Ware and Gasaway (1976) reported increased rough fish populations following removal of vegetation by grass carp *(Ctenopharyngodon idella)*.

In some cases, reduction of aquatic vegetation has coincided with improving fisheries, while others have shown declining fisheries. More research is needed to determine when aquatic plant management will benefit, harm, or have no impact on fisheries. The objective of this study was to assess the immediate impacts of aquatic plant management on the native fish populations in Lake Yale.

Study Area

Lake Yale is a 1,636-ha lake located in Lake County near Eustis, FL. Hydrilla *(Hydrilla verticillata)* became established in the southern end of Lake Yale in 1980, at which time Illinois pondweed *(Potamogeton illinoensis)* covered 140 ha and was the dominant submersed plant. Hydrilla expanded from 1981 to 1984 despite various herbicide treatments that included Aquathol K, Diquat plus copper, and Sonar. Hydrilla (1,400 ha) was reduced (128 ha) after a large-scale Sonar treatment in April 1984, and Illinois pondweed expanded to 320 ha. No more hydrilla was found until 1987 (2 ha), while Illinois pondweed expanded to a peak 1,080 ha, and total plant coverage was 45 percent. Between

[1] Florida Game and Fresh Water Fish Commission, Eustis, FL.

1987 and 1990, 21,145 triploid grass carp (12.9/hectare) were stocked, but hydrilla growth continued. By 1991, hydrilla expanded to 1,340 ha, Illinois pondweed coverage declined to 325 ha, and total plant coverage peaked at 85 percent. Sonar treatments in 1992 and 1993 in addition to triploid grass carp nearly eliminated hydrilla by the end of 1993. By that time, Illinois pondweed declined to 140 ha, and total vegetation coverage was 20 percent (Hestand, Thompson, and Mallison in preparation).

Materials and Methods

The game, forage, and rough fish populations (Table 1) in Lake Yale were evaluated through annual block-net sampling. Nets (0.4 ha) were set in two fixed shoreline stations (depth 0.5 to 3 m) and two fixed offshore stations (depth 3 to 5 m) during May or June each year. Aquatic vegetation was primarily maidencane *(Panicum hemitomon)*, arrowhead *(Sagittaria lancifolia)*, and spikerush *(Eleocharis* spp.*)* in shoreline areas and Illinois pondweed, hydrilla, and southern naiad *(Najas guadalupensis)* in offshore areas. Nets were sampled using rotenone at a concentration of two parts per million. Fish were collected over a 3-day period, identified to species, and individually measured (total length) and weighed. Harvestable game fish were classified as follows: largemouth bass ≥360 mm, black crappie *(Pomoxis nigromaculatus)* ≥240 mm, chain pickerel *(Esox niger)* ≥360 mm, and bluegill *(Lepomis macrochirus)*, redear sunfish *(L. microlophus)*, warmouth *(L. gulosus)*, spotted sunfish *(L. punctatus)*, and redbreast sunfish *(L. auritus)* ≥160 mm.

The Lake Yale largemouth bass population was further investigated through quarterly electrofishing surveys. Twelve fixed shoreline transects were established at the beginning of the study period. Each transect was fished for 15 min (pedal time) in February and August of each year. Largemouth bass were individually weighed, measured, and returned to the lake. Data is reported from spring electrofishing samples (February and May samples combined).

Table 1
Classification of Fish Species Caught During Lake Yale Block-net Sampling Between 1987 and 1993

Game Fish	Forage Fish	Rough Fish
Black crappie	Eustis pupfish	Bowfin
Bluegill	Florida flagfish	Brown bullhead
Chain pickerel	Banded killifish	Yellow bullhead
Largemouth bass	Bluefin killifish	White catfish
Redbreast sunfish	Least killifish	Florida gar
Redear sunfish	Seminole killifish	Longnose gar
Spotted sunfish	Mosquitofish	Atlantic needlefish
Warmouth	Pirate perch	Gizzard shad
	Pugnose minnow	Lake chubsucker
	Redfin pickerel	
	Sailfin molly	
	Golden shiner	
	Taillight shiner	
	Brook silverside	
	Inland silverside	
	Bluespotted sunfish	
	Dollar sunfish	
	Swamp darter	
	Tadpole madtom	
	Threadfin shad	
	Banded topminnow	
	Golden topminnow	

Results and Discussion

Total native fish biomass (four block-net totals) ranged from 105 to 145 kg/hectare in all years except 1988 where 60 kg/hectare were collected (Figure 1). In 1993, biomass of all groups (game, forage, and rough fish) approximated 1987 values. Shoreline nets made up 55 to 60 percent of the sample from 1987 to 1989, 41 percent in 1990, 64 percent in 1991 and 1992, and 86 percent in 1993. Biomass in shoreline nets averaged 122 kg/hectare from 1987 to 1992 (45- to 85-percent vegetation coverage in the lake) and increased by 90 percent to 231 kg/hectare in 1993 (20-percent vegetation coverage in the lake). Biomass in offshore nets averaged 95 kg/hectare from 1987 to 1992 and declined by 60 percent to 38 kg/hectare in 1993.

Game fish biomass ranged from 79 to 119 kg/hectare except in 1988 where it dropped to 44 kg/hectare. Game fish biomass in shoreline

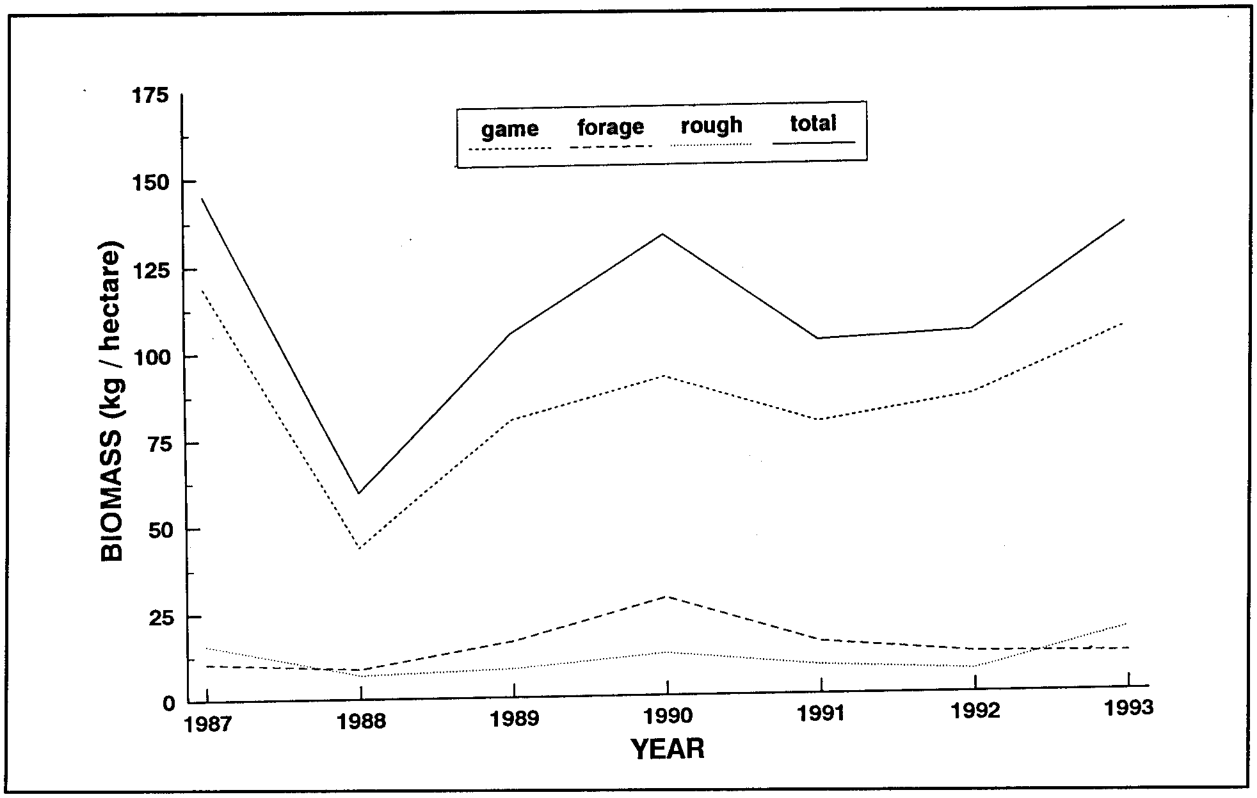

Figure 1. Lake Yale block-net results for biomass of game, forage, and rough fish from 1987 to 1993

and offshore nets revealed similar trends until 1993, when 89 percent of the sample was collected in shoreline nets (Figure 2A). Game fish biomass shifted to vegetated shoreline areas in 1993, when submersed plants were greatly reduced in offshore areas. Offshore game fish biomass peaked in 1990 at 112 kg/hectare (61 percent of game fish biomass). Biomass of harvestable game fish showed a similar pattern as offshore biomass peaked in 1987 and 1990 at 30 kg/hectare (56 percent of the samples), while 94 percent of the 1993 sample was collected in shoreline nets (Figure 2B).

Bluegill dominated game fish biomass at an average 38 kg/hectare and 40 to 50 percent of game fish biomass (Figure 3). Redear sunfish was next at an average of 16 kg/hectare and 15 to 30 percent of game fish biomass. Largemouth bass biomass averaged 11 kg/hectare and made up 10 to 15 percent of game fish biomass. There was an overall decline in the numbers and biomass of all harvestable game fish except chain pickerel (Figure 4). Redear sunfish was the dominant harvestable game fish and averaged 41 percent of the biomass and 45 percent of the number. Harvestable redear sunfish biomass peaked at 16 kg/hectare (88 fish/hectare) during peak hydrilla growth (1991) and declined to 5 kg/hectare (28 fish/hectare) in 1993. Bluegill was the second most common harvestable game fish by biomass and number, but declined to less than 1 kg/hectare (2 fish/hectare) in 1993. Chain pickerel increased substantially in 1993 and dominated harvestable biomass at 43 percent (29 percent of the number), while redear sunfish remained most abundant at 40 percent of the number of harvestable game fish (28 percent of the biomass).

Total biomass of largemouth bass declined over the study from 16 kg/hectare in 1987 to 11 kg/hectare in 1993, and the number of fish declined by 43 percent (Figure 4). From 1987 to 1990, most of the largemouth bass were captured offshore, while shoreline nets contained the majority of the sample in 1991 through 1993 (Figure 5). In 1993, 80 percent of the sample was collected in shoreline nets. The largemouth bass, like other game fish species, moved into vegetated shoreline areas.

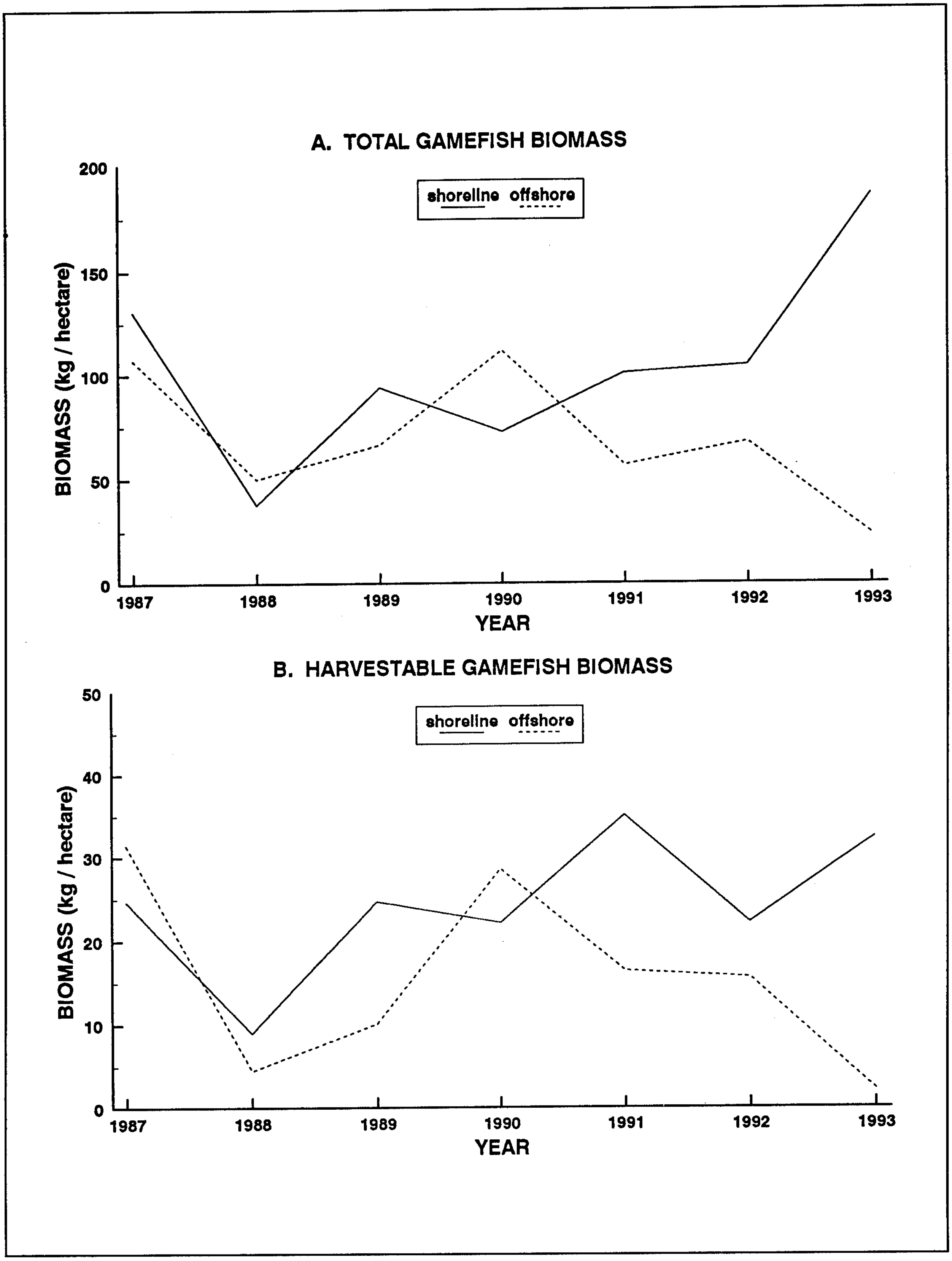

Figure 2. Lake Yale block-net results for biomass of (A) total and (B) harvestable game fish (largemouth bass, black crappie, chain pickerel, redear sunfish, bluegill, spotted sunfish, and warmouth) from 1987 to 1993

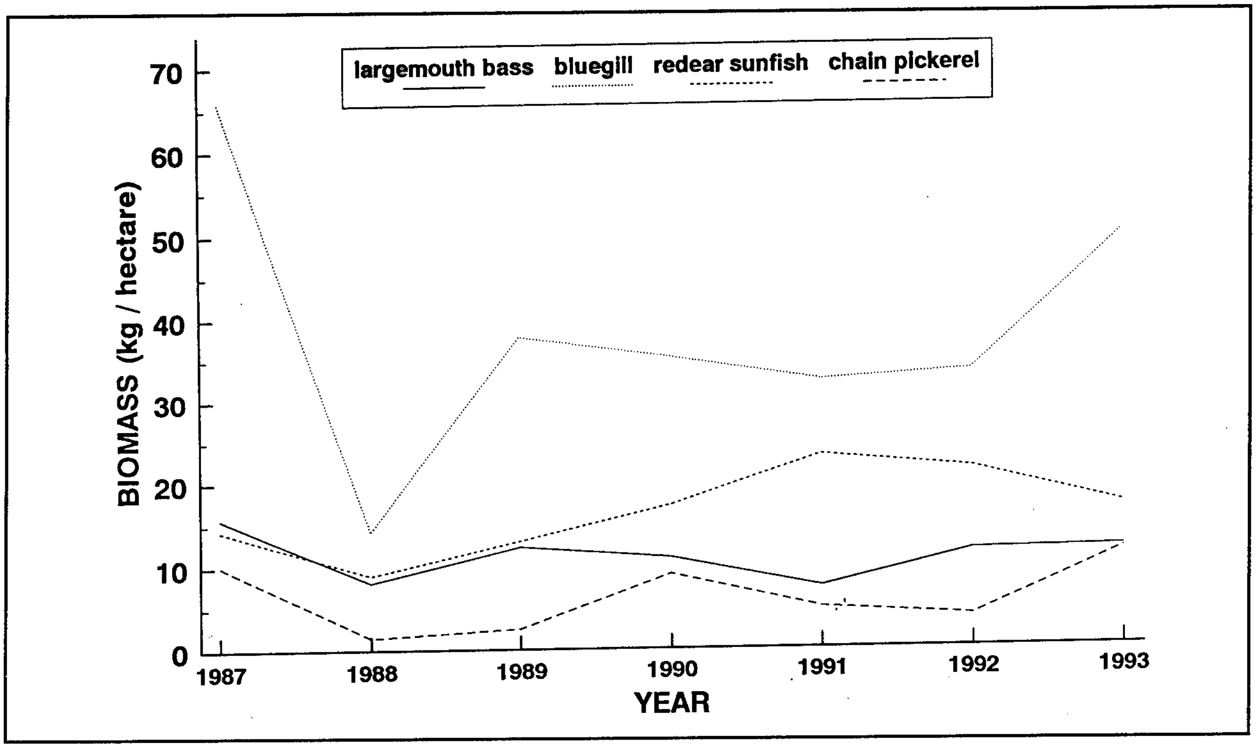

Figure 3. Lake Yale block-net results for biomass of major game fish from 1987 to 1993

The number of harvestable largemouth bass dropped from six fish/hectare (7 kg/hectare) in 1987 to four fish/hectare (3 kg/hectare) in 1993.

Electrofishing results for largemouth bass indicated that recruitment (number less than 200 mm total length) was highest in 1988 (peak Illinois pondweed coverage) and 1990 to 1992 (peak hydrilla coverage) (Table 2). Electrofishing revealed increases in the number of quality-size (greater than 300 mm total length) and harvestable fish, proportional stock density (PSD), relative stock density (RSD), and relative weight (W_r) (Anderson and Gutreuter 1983). The largemouth bass fishery improved from 1987 to 1990 as the number of quality-size fish doubled, PSD increased from 37 to 45 percent, the number of harvestable fish increased by more than 50 percent, and RSD increased from 10 to 14 percent (Table 1). By 1993, the number of quality-size fish increased to 142, PSD increased to 62 percent, the number of harvestable fish nearly doubled again to 70, and RSD increased to 30 percent. Relative weight improved slightly from 86 to 89 over the study period, possibly a result of increased availability of forage fish because of less submersed aquatic plants in 1992 and 1993 (Bailey 1978; Colle and Shireman 1980; Savino and Stein 1982). Although electrofishing may give reliable estimates of largemouth bass density (Hall 1986), the higher numbers of large fish in the 1993 Lake Yale sample reflect increased susceptibility to the sampling gear as fish have moved into vulnerable shoreline areas. The increased number of harvestable largemouth bass in 1993 samples (five-fold increase in electrofishing and four-fold increase in blocknets since 1992) may have been influenced by reduced adult mortality resulting from the establishment of a minimum-length limit at 356 mm on July 1, 1992. In addition, the dominant year classes (1990 and 1991) reached harvestable and quality size in 1993. The lowest number and biomass of harvestable largemouth bass were collected (electrofishing and block-nets) during peak hydrilla coverage (1991 and 1992). Young-of-the-year survival was correlated with increasing levels of hydrilla in 1990 and 1991, as observed in numerous other studies (Ware and Gasaway 1976; Moxley and Langford 1982; Colle et al. 1987; Klussman et al. 1988; Porak et al. 1990).

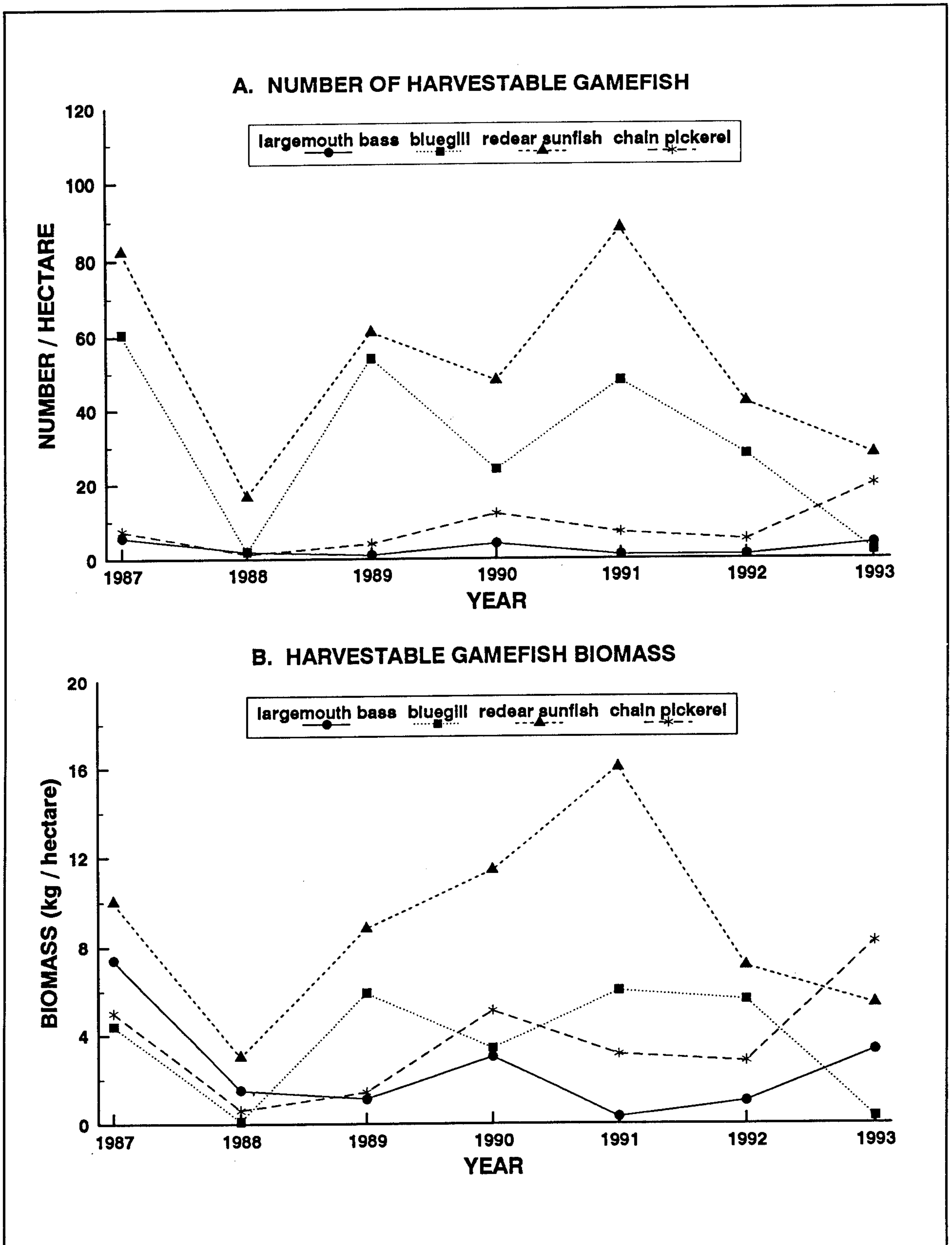

Figure 4. Lake Yale block-net results for (A) number and (B) biomass of harvestable game fish from 1987 to 1993

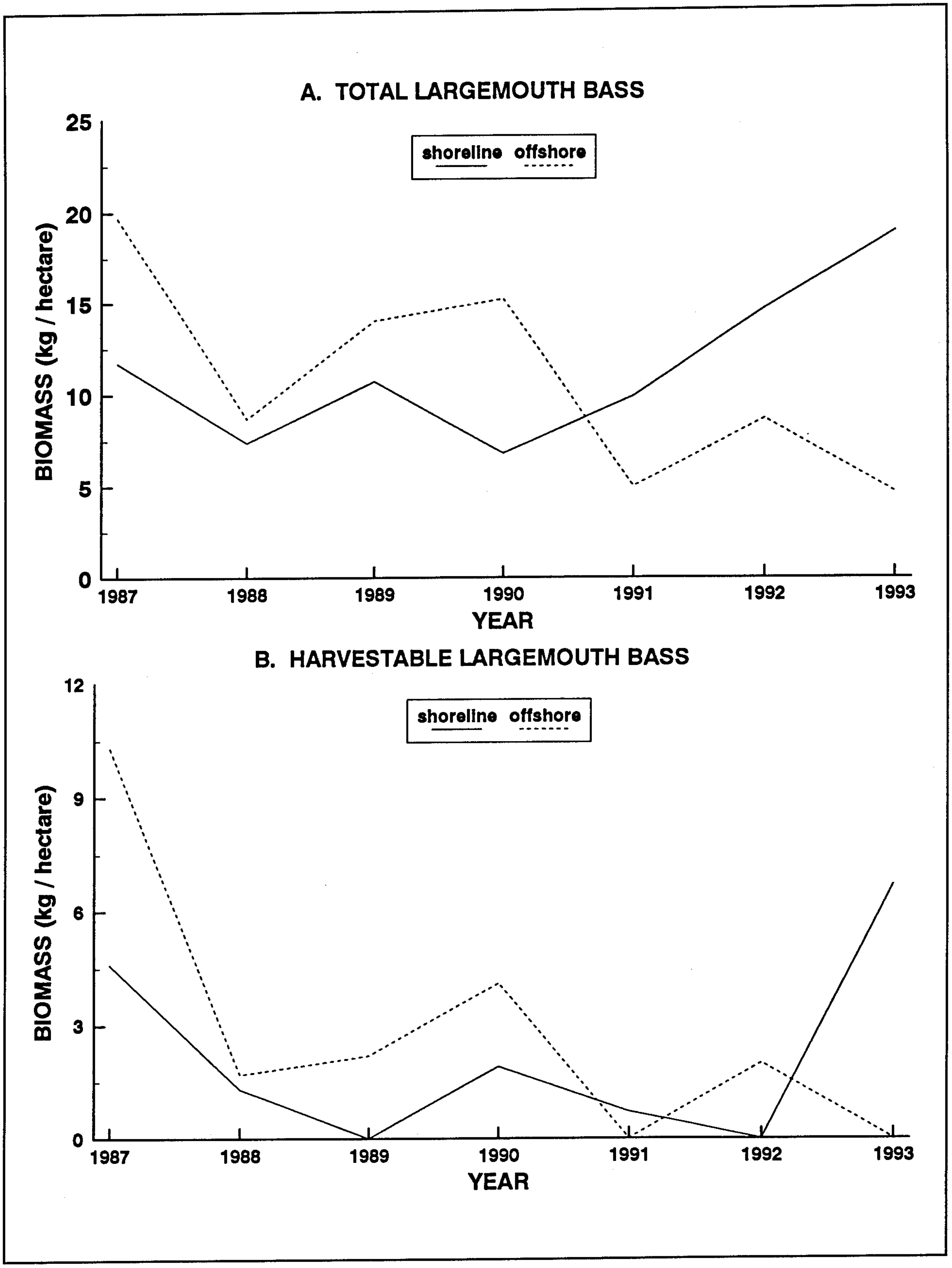

Figure 5. Lake Yale block-net results for biomass of (A) total and (B) harvestable largemouth bass from 1987 to 1993

Table 2
Lake Yale Spring Electrofishing Results (February and May samples combined, total 6 hr per year) for Largemouth Bass, 1987-1993

Year	Number of Fish	Mean W_r	Recruitment (No. <200 mm)	No. Quality (PSD)	No. Harvestable (RSD)
1987	236	86	72	60 (37%)	17 (10%)
1988	422	88	162	75 (29%)	22 (9%)
1989	368	87	98	145 (54%)	35 (13%)
1990	376	86	110	119 (45%)	36 (14%)
1991	407	85	172	82 (35%)	16 (7%)
1992	238	87	113	56 (45%)	13 (10%)
1993	321	89	91	142 (62%)	70 (30%)
Mean	338	87	117	97 (44%)	30 (13%)

Forage fish biomass peaked at 28 kg/hectare (21 percent of total block-net biomass) in 1990, when hydrilla biomass increased by 150 percent (Figure 1). In 1991, hydrilla expanded by 250 percent and forage fish biomass declined to 15 kg/hectare. Forage fish biomass ranged from 9 to 16 kg/hectare in all other years and was 11 kg/hectare in the beginning and ending years. The number of forage fish collected in 1993 was more than twice that of 1987, illustrating a greater abundance of small fish (primarily cyprinids) at less than half the vegetation coverage. In 1993, 74 percent of forage fish biomass was collected in shoreline nets (compared with 63 percent in 1987).

In 1989 and 1990, 70 percent of the forage fish biomass was collected in offshore nets and was dominated by bluespotted sunfish *(Enneacanthus gloriosus)* at 44 percent of the sample. Golden shiners *(Notemigonus chrysoleucas)* were next with 18 percent of the biomass in 1989 and 1990. In other years, dollar sunfish *(Lepomis marginatus)*, Seminole killifish *(Fundulus seminolis)*, and bluefin killifish *(Lucania goodei)* dominated at a combined 50 to 60 percent of the forage biomass. In 1993, dollar sunfish comprised 33 percent of the sample and increased by 68 percent over the study period, while bluespotted sunfish diminished to less than 1 kg/hectare. Golden shiner biomass ranged from 10 to 20 percent of the forage base (1 to 6 kg/hectare) throughout the study. Threadfin shad *(Dorosoma petenense)*, with biomass less than 1 kg/hectare, have been insignificant (less than 5 percent of the sample) except for 1987 where 2 kg/hectare (18 percent of forage fish biomass) were collected.

Biomass of rough fish ranged from 7 to 16 kg/hectare over the study period, peaking in 1993 at 18 kg/hectare and 14 percent of total block-net biomass (Figure 1). More than 75 percent of the rough fish sample was taken in shoreline nets in all years. Bowfin *(Amia calva)* was the dominant rough fish from 1987 to 1989 at 3 to 7 kg/hectare, but declined by 38 percent over the study. Lake chubsucker *(Erimyzon sucetta)* was the dominant rough fish from 1990 to 1993 at 4 to 10 kg/hectare and has increased by 238 percent since 1987.

Conclusions

Aquatic plant coverage in Lake Yale fluctuated from 45 to 85 percent between 1987 and 1992 and declined to 20 percent in 1993. No immediate impact on total native fish biomass was observed. However, shoreline nets contained twice the biomass and offshore nets less than half the biomass in 1993 (20-percent vegetation coverage) than the averages of all other years (45- to 85-percent vegetation coverage). The number and biomass of harvestable game fish (except chain pickerel) declined during the study, and total harvestable biomass was 15 percent less in 1993 than the average of all other years. This could not be directly attributed to the changes in the vegetation since biomass began to decline before removal of submersed vegetation. Game fish migrated to vegetated shoreline areas in 1993 in response to the declining

aquatic plant community in offshore areas. Forage fish biomass was the same in the beginning and ending years, while the number of forage fish in 1993 was more than twice the 1987 number. Rough fish biomass increased by 15 percent over the study period.

Largemouth bass biomass and number (total and harvestable) collected by electrofishing declined over the study period, and the lowest catch samples of harvestable largemouth bass were collected during peak hydrilla coverage. There was a positive correlation between recruitment of largemouth bass and aquatic vegetation coverage. The number of quality-size and harvestable fish, PSD, RSD, and relative weight collected by electrofishing was highest in 1993, when submersed vegetation coverage was lowest (20 percent). These improvements were attributed primarily to migration of largemouth bass into vulnerable shoreline areas (more susceptible to the gear) and, to a lesser extent, to improved foraging success with vegetation removal, recruitment of dominant year classes to quality and harvestable sizes in 1993, and reduced adult mortality resulting from a minimum length limit imposed in 1992.

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Long-Term Use of Grass Carp for Aquatic Plant Control in Deer Point Lake, Bay County, Florida

by
Jess M. Van Dyke[1]

Because the beautiful beaches of the Gulf of Mexico are magnets for tourist development, most of the rapid population growth in the Florida Panhandle has been coastal. The growth boom beginning in the 1950s caused a decline in groundwater resources near coastal areas. This problem was particularly acute in the Panama City Area, where groundwater levels had dropped 200 ft and saltwater intrusion threatened by the late 1950s. The "city fathers" realized that fresh water was fast becoming the limiting factor to growth. Therefore, in 1961, a dam was constructed across North Bay at a constriction where deer often swam. Deer Point Lake was formed by a low-head, causeway dam fixed at 4.5 ft MSL. The 1,200-acre estuary with "an extensive *Juncus* marsh" became a 4,700-acre, freshwater lake (Crittenden 1957). Six hundred million gal a day of pure, fresh water was available as a municipal and industrial water supply, twenty times more than Bay County could use. The Panama City area was the only area in Florida where groundwater became more abundant after 1961 (Fernald and Patton 1984). Real estate interests sprang at the opportunity to develop the shore of the new reservoir.

The inundated estuary plotted its revenge for 7 years. Then in 1968, John Crew of the Florida Game and Fresh Water Fish Commission (GFC) noted that native, aquatic plants were becoming "troublesome" (Crew 1968). Some time after 1972, the exotic Eurasian watermilfoil (*Myriophyllum spicatum*) was introduced into Deer Point Lake. By 1975, according to Jerry Krummrich of GFC, emersed species occupied 19 percent of the lake, while submersed species, mostly a mixture of Eurasian watermilfoil and Illinois pondweed (*Potamogeton illinoensis*), occupied 23 percent of the system (Krummrich 1975). Aerial photography by the Department of Natural Resources (now Department of Environmental Protection) revealed that the submersed vegetation reached the surface of the lake in a near shoreband at least 100 yd wide. Boating access was blocked, and an airboat was required to navigate the system.

Blocking recreational access was a problem, but obstructing the municipal water supply was a crisis. Something had to be done, and the large-scale use of herbicides was not a viable option in this potable water supply. The large size of the system and the abundance of flooded timber negated the use of mechanical harvesting. Furthermore, the lake could be drawn down only approximately 3.5 ft. The political pressure on Dr. Alva P. Burkhalter, the Chief of the Department's new Bureau of Aquatic Plants, was extreme. Meanwhile, from Arkansas was coming good news about biological control using grass carp, which "do not strip lakes of all cover, but leave upright, emergent swamp grasses, reeds, and lilies to serve as cover" (Bailey 1975). At that time, Arkansas was the only state where grass carp had been used to control aquatic plants in large reservoirs. Because Deer Point Lake flowed "into a saltwater bay," Dr. Burkhalter proposed the use of this fish without containment to control the lake's severe submersed vegetation problem. The Department asserted that it should determine where to stock grass carp in Florida, stating it had authority over aquatic plant control; the GFC balked saying it had authority over fish. The major concerns at that time were escapement, reproduction, and naturalization of grass carp. Those concerns

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were shared by the field biologists of both agencies. Ultimately, the GFC successfully challenged in court the Department's authority to release grass carp into Deer Point Lake on the grounds that the fish could escape over the dam. The issue was so political, however, that even after the Commission established its authority over grass carp, it agreed to stock Deer Point Lake.

With the flap-gate doors closed, the dam may have made a fairly effective fish barrier; but during heavy rains, the doors were open, and fish could easily escape into North Bay. Fortunately, salinity there averages about 20 ppt, which is lethal to grass carp. In North Bay, the maximum distance grass carp were found from the dam was 9 miles (Hardin 1980). A fish barrier was constructed across the entire dam in 1988. Nevertheless, escaped grass carp can still be found congregating in the fresh water just below the dam.

From October 1975 to August 1977, 107,500 grass carp fingerlings were stocked into grow-out embayments, then released into the main body of Deer Point Lake. The Executive Director of the Department of Natural Resources at that time, Harmon Shields, said, "If the grass carp don't work, we will have to remove the vegetation by hand." His field biologists understood that "we" did not mean "him" either. In order to ensure success, an additional 10,000, 1-lb grass carp were stocked in November and December of 1978. The goal was "full, rapid weed control" (Van Dyke 1979).

Since September of 1974, the Department's field biologists had been monitoring vegetation transects on Deer Point Lake. That last stocking of 10,000 grass carp gave them some apprehension. According to the transect data, grass carp were selecting the Illinois pondweed from stands mixed with Eurasian watermilfoil. A classic response of a nonpreferred target plant was unfolding. As the preferred pondweed was selectively consumed by the grass carp, the watermilfoil actually increased, then declined only after the preferred species disappeared from the transects (Figure 1).

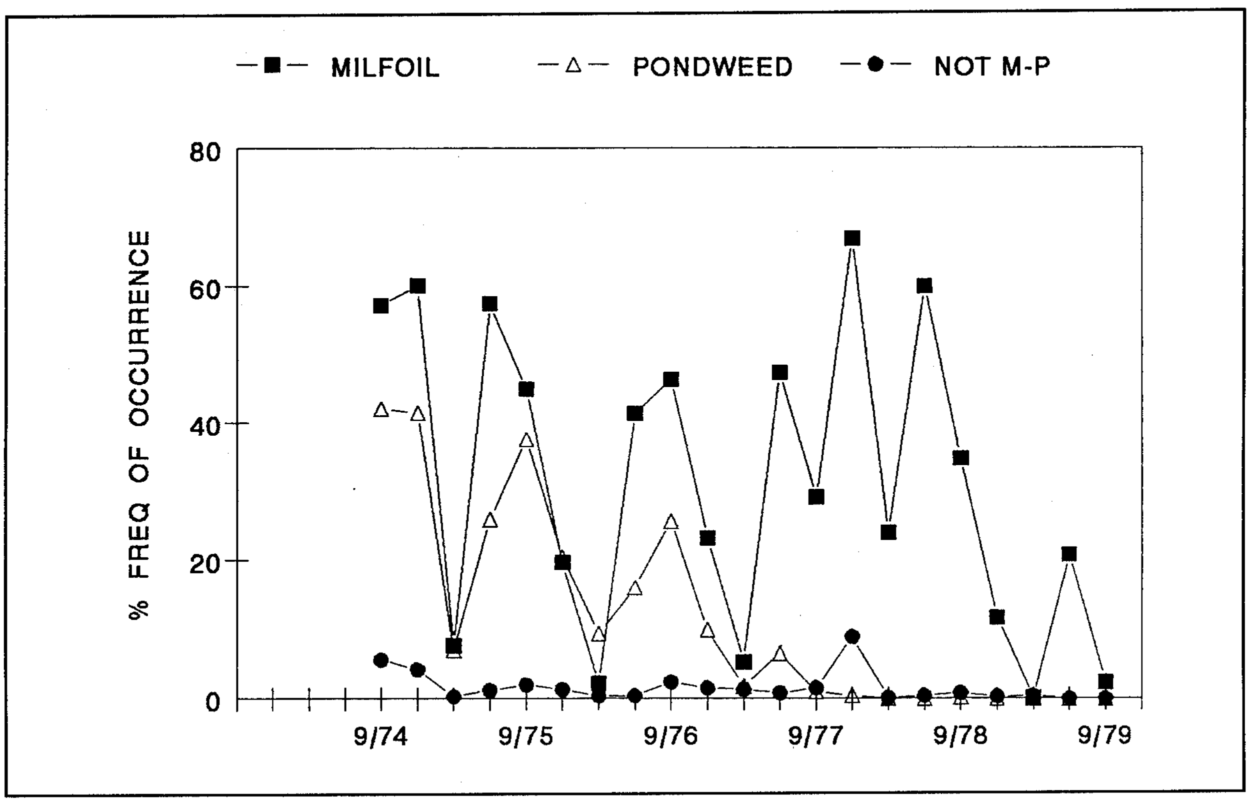

Figure 1. Deer Point Lake vegetation transect data from 1974 to 1979 (DEP)

In the late 1970s and early 1980s, the aquatic plants were controlled in Deer Point Lake from the stocking of 117,500 grass carp (25/acre). Boating access improved dramatically. Most importantly, there was no obstruction to the municipal water supply intake. Everything was not perfect though. In time, not only the submersed species but most of the emersed species were greatly reduced. Large areas of cattails (*Typha* sp.), fragrant water lily (*Nymphaea odorata*), and dollar-bonnet (*Brasenia schreberi*) were eliminated. Shoreline erosion increased because of the decline in emersed vegetation, so many seawalls were constructed.

In 1982, the field biologists of the Department and the GFC ended the practice of conducting separate vegetation transects and began working together, thanks to the cooperative spirit of GFC's Norman Young. Three years later, Eurasian watermilfoil came back with a vengeance, but not the preferred natives, in another classic response to grass carp food preference (Figure 2). The water intake was becoming obstructed again. Therefore, in October of 1985, 40,000 triploid grass carp (8.5/acre) were stocked into Deer Point Lake. The next month, Hurricane Kate destroyed the center section of a fish barrier that had been erected in front of the drawdown structure, and the structure was opened to reduce flooding. Though some grass carp were probably lost, aquatic plant control was again achieved and maintained until 1993. At that point, interesting plant responses began to occur. Nonpreferred native plants, particularly lemon bacopa (*Bacopa caroliniana*) began to appear in abundance. Eel grass (*Vallisneria americana*) also began to expand. The expansion of nonpreferred native plants was considered beneficial in that some habitat for fish and waterfowl was again available. Long term, it is possible to use grass carp to greatly reduce an aggressive exotic without the elimination of all native vegetation, but there will be a shift to nonpreferred native species.

The recent resurgence of vegetation in Deer Point Lake can also be attributed to the expansion of native species that are preferred by the grass carp, such as Southern naiad

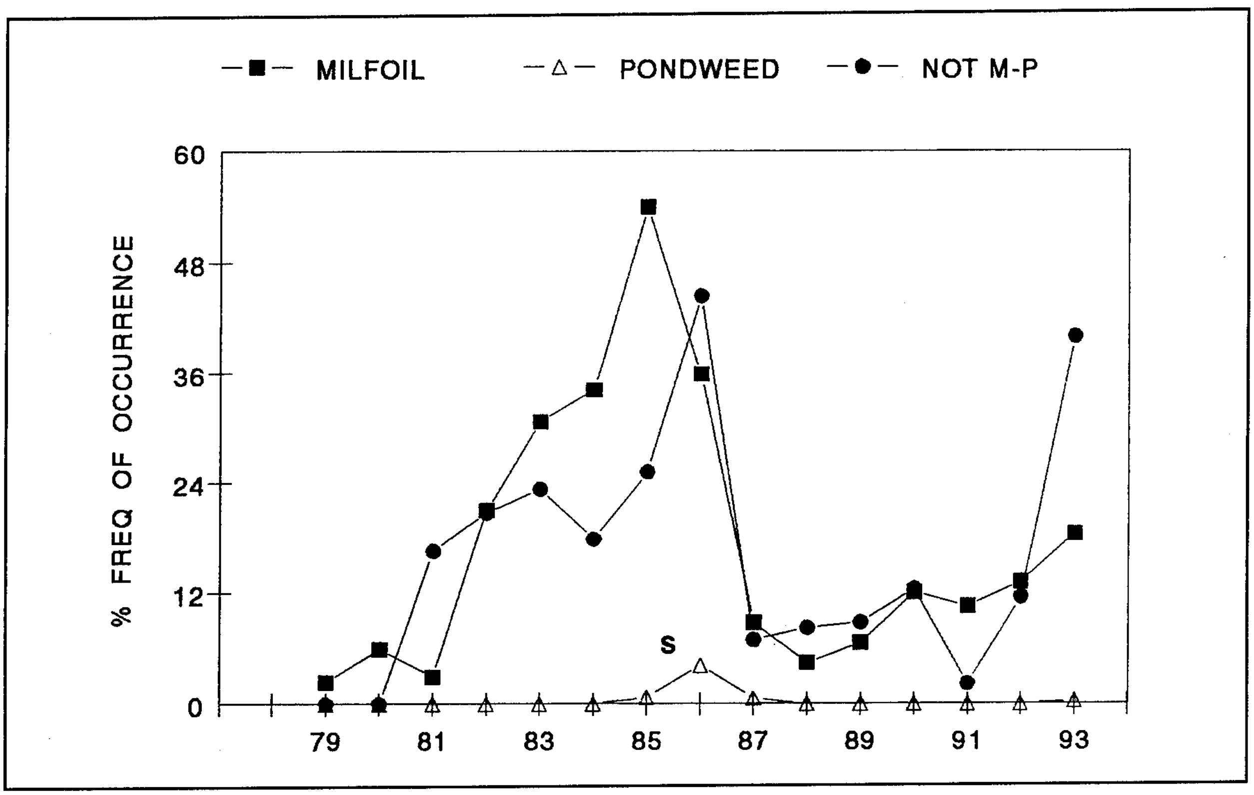

Figure 2. Deer Point Lake vegetation transect data from 1979 to 1993 (GFC/DEP)

(*Najas guadalupensis*) and stonewort (*Nitella* spp.). The presence of these preferred species was cause for alarm because it indicated that the feeding activity of grass carp, last stocked in 1985, was insufficient to protect the lake from hydrilla (*Hydrilla verticillata*). Hydrilla is a far greater threat to Deer Point Lake than Eurasian watermilfoil, and the range of hydrilla is rapidly expanding across the Florida Panhandle (Figure 3). It is likely that grass carp have already prevented the establishment of hydrilla inadvertently introduced into Deer Point Lake via boat trailers. To maintain this protection and to avoid nuisance levels of other aquatic plants, 10,000 triploid grass carp were stocked in Deer Point Lake in March 1994. The goal of this rather low stocking rate (two grass carp/acre) is to provide some suppression in the upsurge of submersed species while preventing the establishment of hydrilla.

In summary, there are lessons learned in 20 years of monitoring the effects of grass carp in Deer Point Lake. The first is that political and economic interests want quick results with the grass carp. Overcontrol from a fish and wildlife standpoint often results, and overcontrol of aquatic vegetation tends to promote shoreline development. The second is that field biologists from different agencies working together to collect and interpret data can produce a consensus that has the power to steer policy in a direction more protective of a lake's ecosystem. The third lesson is that because the weather is unpredictable and each lake is unique, determining the correct stocking rate of grass carp for a given lake is an art, not a science. Models are only guidelines. The fourth lesson is that, though we are all understaffed and underfunded, the long-term monitoring of aquatic plant control using grass carp is crucial. Finally, the grass carp may be viewed as a herbicide with unique characteristics. The grass carp "herbicide" is very inexpensive ($5 to 10/acre/year); it is slow release (obvious results may take several years); it is very persistent (effects may last 10 to 20 years); it is highly mobile; it is moderately selective, and its selectivity varies through time; and it is highly effective though somewhat unpredictable. The grass carp "herbicide" should generally be used at low rates as a supplemental or follow-up treatment to some other aquatic plant control technique. Avoid "drift"; use grass carp in contained areas only. All other herbicides have basic

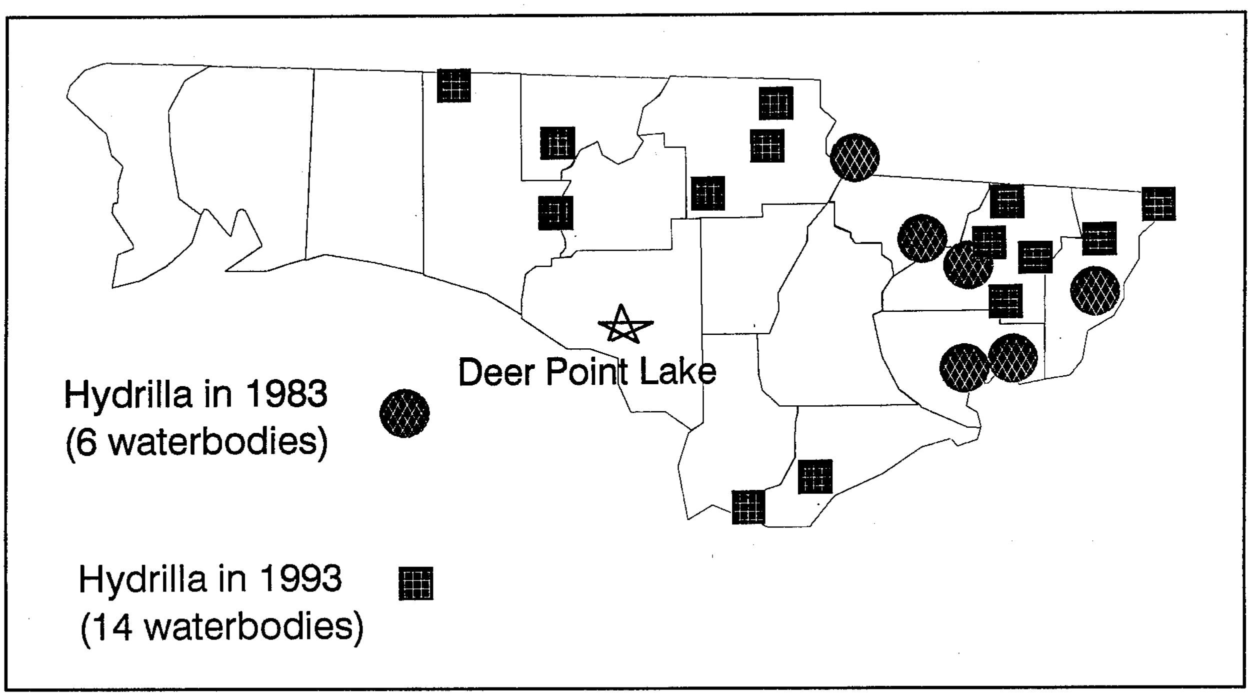

Figure 3. The expansion of Hydrilla verticillata in the Panhandle of Florida (1983-1993)

Federal guidelines. The grass carp "herbicide" is more powerful and persistent than any chemical labeled for aquatic use. Because of the trend in many States of using grass carp without containment, it is clear that Federal "use restrictions" on grass carp are needed to protect valuable fish and wildlife habitat.

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Economic Considerations of Integrated Hydrilla Management A Case History of Johns Lake, Florida

by
Bruce V. Jaggers[1]

Introduction

Aquatic plant managers in the field make recommendations based largely on biology; however, actual management decisions are many times based largely or entirely on economics and politics. Human population growth in Florida has put tremendous pressure on water resources with dire biological, economic, and political consequences (Stocker 1993). Since 1980, the State of Florida has spent $70,000,000 on hydrilla (*Hydrilla verticillata*) control in public water bodies alone, an average of $4.67 million/year.[2] In Florida, there has recently been a growing gap between funding needs and funding available for hydrilla management, and the current funding gap (fiscal year 1994-95) stands in excess of $6,000,000.[2] Some lakes have been denied hydrilla treatment in the last few years because of funding shortfalls, and canals have basically not been funded the last 2 years because of the same funding shortfalls. Phillippy et al. (1989) found hydrilla established in 59 of the 100 largest lakes in Florida and calculated that a 10-percent (total surface area) fluridone herbicide treatment to control hydrilla in those 59 lakes would cost approximately $48,000,000. By 1992, hydrilla was known to be established in 69 of these same 100 largest lakes (Trent et al. 1992). If the remaining 31 largest lakes became infested with hydrilla, the economic consequences would be unmanageable at current funding levels. Funding levels have been decreasing, not increasing, in recent years.[2] The specter of the inability to afford hydrilla control when necessary translates to loss of biodiversity because of the ability of hydrilla to outcompete native macrophytes (Haller and Sutton 1975; Small, Richard, and Osborne 1985; Langeland 1990; Howard-Williams 1993), increased sedimentation rates (Joyce et al. 1992), stunted and slow-growing game fish populations (Hinkle 1986; Langeland 1990; Killgore, Hoover, and Morgan 1991), impairment of multiple uses (Small, Richard, and Osborne 1985; Langeland 1990), and loss of economic revenue (Colle et al. 1985, 1987; Langeland 1990). Maximum benefits to fish and wildlife attributed to macrophyte populations generally occur at intermediate macrophyte cover and density (Hinkle 1986; Wiley et al. 1984; Wiley, Tazik, and Sobaski 1987; Langeland 1990; Killgore, Hoover, and Morgan 1991; Petr 1987, 1993).

The use of grass carp alone or integrated with other hydrilla management methods has proven to be an efficient and cost-effective management tool (Stott et al. 1971; Stott and Buckley 1978; Osborne 1982; Shireman, Colle, and Canfield 1986; Sutton, Vandiver, and Neitzke 1986; Sutton and Vandiver 1986; Leslie et al. 1987; Wiley, Tazik, and Sobaski 1987; Pershe, Setaram, and Baird 1987; Cooke 1988) and may need to be increasingly relied on in the future if economic downturns continue. However, one fear of using integration is that this technique can result in nontarget effects because of its sometimes unpredictable power or unpredicted climatic events. This report accounts for a case history of a hydrilla control program in Johns Lake, Florida,

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[2] Personal Communication, 1994, Jeff D. Schardt, Florida Department of Natural Resources, Tallahassee, FL.

integrating triploid grass carp (*Ctenopharyngodon idella*) and contact herbicides. A brief economic analysis comparing this technique with a program using fluridone herbicide for hydrilla control is included along with largemouth bass (*Micropterus salmoides*) response to the habitat alteration.

Materials and Methods

Johns Lake is a 979-ha lake located in central Florida at latitude 28°32′ and longitude 81°38′ and is the 64th largest named lake in Florida (Shafer et al. 1986). Substrate is generally sandy with limited accumulations of silt. Some limited organic accumulation occurs in deeper water zones. Johns Lake is a prairie-type lake, and water levels have fluctuated over a range of approximately 4 m since 1960. During 1981, a drought year, large expanses of maidencane (*Panicum hemitomon*) became established in offshore areas, creating morphologically diverse habitat. The number of macrophyte species has also been diverse in recent years, with approximately 14 submersed and over 20 emergent and floating-leaf macrophyte species represented. Johns Lake has historically been utilized primarily for fishing, with very little other multiple-use activity occurring. Hydrilla was first noted in 1985 and was well established (0.8 ha); contact herbicide treatments targeting hydrilla were initiated in January 1986 (Table 1). Triploid grass carp were integrated beginning February 1988 and stocked over a period of 3 years (Table 2) to keep a population of small fish (25.4 to 43.5 cm) present in the lake, as unpublished data and project observations indicated that smaller fish targeted hydrilla more effectively. Also, serial stocking is indicated to be more efficient and cost-effective over the long run (Wiley, Tazik, and Sobaski 1987). Integrated control with contact herbicides was selected because of the possibility of funds not being available for large-scale fluridone herbicide treatments. An economic analysis was made by comparing the cost of materials (triploid grass carp and contact herbicides) actually used in this management program with what the cost of materials would have been (contact herbicides and fluridone) in a strictly herbicidal management program. The projected costs of a fluridone hydrilla management program were based on conservative assumptions of a full-lake treatment first being required in 1988 and repeat treatments being required once every 3 years. Cost calculations were based on $1,000/3.8 L of fluridone product and an 8.3-percent (81.2-ha) treatment at a rate of 4.7-L fluridone/hectare treated. The objective of this hydrilla management program in Johns Lake was to reduce or eliminate hydrilla while maintaining as much native vegetation as possible. The west side of Johns Lake was surveyed biannually using a recording fathometer (Raytheon model DE-719C) to determine percent frequency of occurrence and percent volume infestation of macrophytes in the water column as described by Maceina and Shireman (1980). The east side of Johns Lake was surveyed and macrophytes qualitatively described every other year by a

Table 1
Contact Herbicide Formulations, Rates, and Area Treated by Year in Johns Lake Integrated Hydrilla Control Program

Date	Formulation	Rate	Hectares Treated
1986	Diquat and Copper	22 and 42 L/ha, respectively	1.6
1987	Aquathol K	56 to 84 L/ha	35.6
1988	Aquathol Granular	359 to 448 kg/ha	31.8
1989	Aquathol Granular	359 to 448 kg/ha	25.5
Total			94.5

Table 2
Number of Triploid Grass Carp Stocked by Season and Resulting Total Stocking Rate per Hectare and Total Stocking Rate per Vegetated Hectare in Johns Lake Integrated Hydrilla Management Program

Date	Number of Triploid Grass Carp Stocked	Total Rate/Hecare	Total Rate/ Vegetated Hectare
Spring 1988	5,000	5.1	7.1
Fall 1988	510	5.6	7.9
Spring 1989	2,000	7.7	9.4
Fall 1989	1,631	9.5	11.4
Spring 1990	1,253	10.6	11.4
Total	10,394		

Department of Natural Resources biologist. Largemouth bass were sampled by biannual electrofishing surveys and total abundance, catch per unit effort (CPUE), proportional stock density (PSD), relative stock density (RSD), and relative weight values (W_r) calculated. A Mini-Bass Small Boat Tournament Trail angler survey from a November 1993 event was also summarized.

Results and Discussion

Macrophytes

Initial hydrilla growth was exponential, even with the concurrent herbicide treatments. However, after stocking of the final group of triploid grass carp in 1990 and expanded herbicide treatments in 1988 and 1989, hydrilla standing crop was eliminated by fall 1991 (Figure 1), with only a trace of hydrilla noted fall 1993. Limited sampling for tubers in the substrate revealed none; therefore, hydrilla tuber reserves in Johns Lake may have been depleted in a span of approximately 3 years. However, tubers are not evenly distributed, and the limited substrate core sampling could have easily missed some concentrations of tubers.[1] However, data from four other central Florida lakes with sandy substrates also indicated probable tuber depletion approximately 3 years after hydrilla standing crop was eliminated (Trent et al. 1992). Data from Rodman Reservoir, Florida, indicated that sandy substrates contained higher numbers of hydrilla tubers than organic substrates. However, the sandy substrate tubers appeared to germinate at a faster rate with multiple dewaterings (Haller and Shireman 1983). Van and Steward (1990) found monoecious hydrilla tubers to survive over 4 years in undisturbed sediments in the

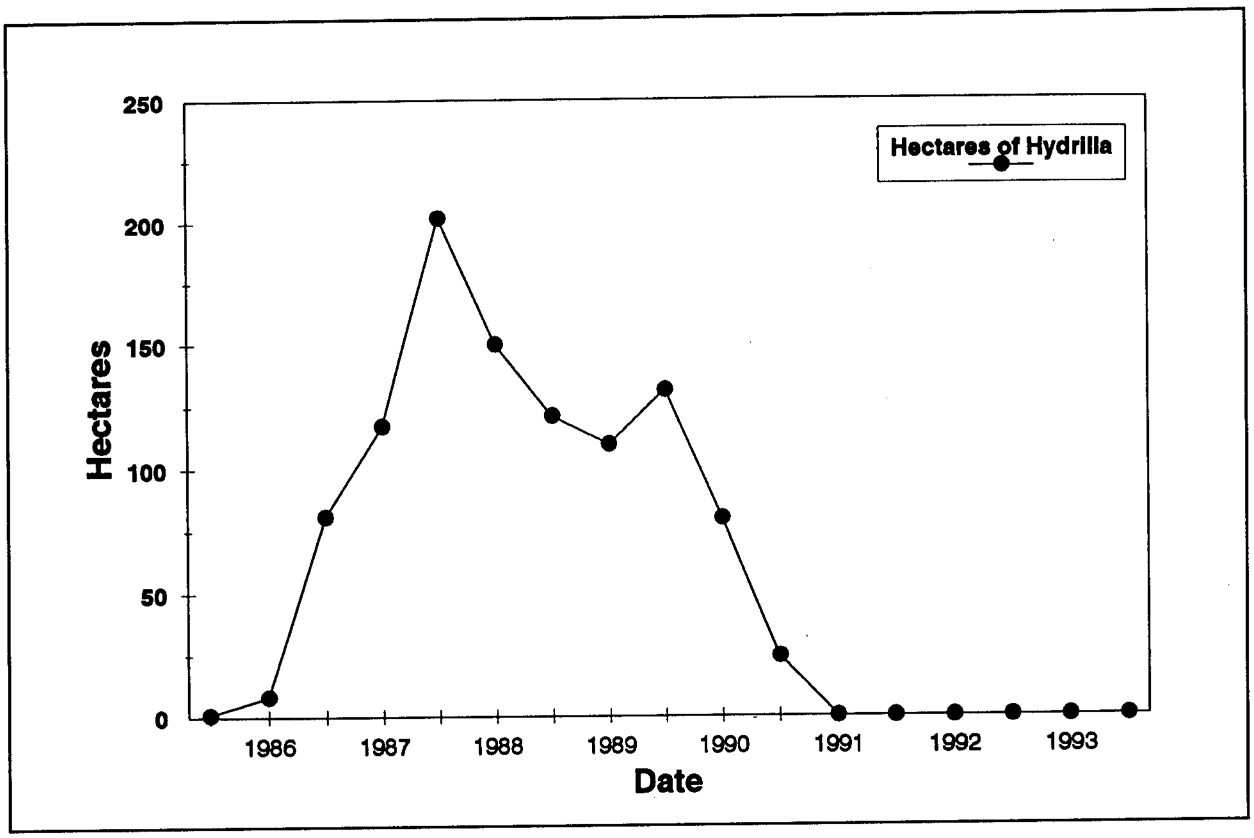

Figure 1. Aerial coverage of hydrilla (hectares) in west Johns Lake by year

1 Personal Communication, 1994, William T. Haller, Center for Aquatic Plants, University of Florida, Gainesville, FL.

laboratory. Depletion of hydrilla tuber and turion propagules in the substrate has not normally been possible using herbicides only. However, new technology utilizing fluridone or bensulfuron methyl to inhibit hydrilla tuber and turion production may soon be available (MacDonald et al. 1993; Langeland 1993; Haller, Fox, and Hanlon 1992; Van and Vandiver 1992).

Concurrent with hydrilla reductions, total native macrophyte coverage in west Johns Lake initially increased and then declined (Figure 2). This data also demonstrates the movement of triploid grass carp through a boatable canal into a nontarget area (northwest pool), as total submersed macrophyte coverage decreased there as well. Percent volume infestation of macrophytes in the water column in west Johns Lake obtained a maxima of 5 to 7 percent in 1988-1989 and then decreased to current values of less than 1-percent volume. Some individual macrophyte species showed increasing trends in coverage (Table 3) such as fragrant water lily (*Nymphaea odorata*), bladderworts (*Utricularia* spp.), and stoneworts (*Nitella* spp.). However, preliminary 1994 data indicate a current decrease in fragrant water lily coverage because of the previous expansion into habitats that will not support this plant over the long-term.[1] Also, major decreases were seen in the emergent offshore community consisting of maidencane and, to a lesser extent, lake rush (*Fuirena scirpoidea*). The current coverage of both these species is less than 1 ha. Offshore maidencane communities did not historically exist during a high water cycle (1960-1973), and rising water levels during 1991 and 1992 probably were responsible for decreased maidencane coverage in some of the deeper water areas. However, most of the maidencane

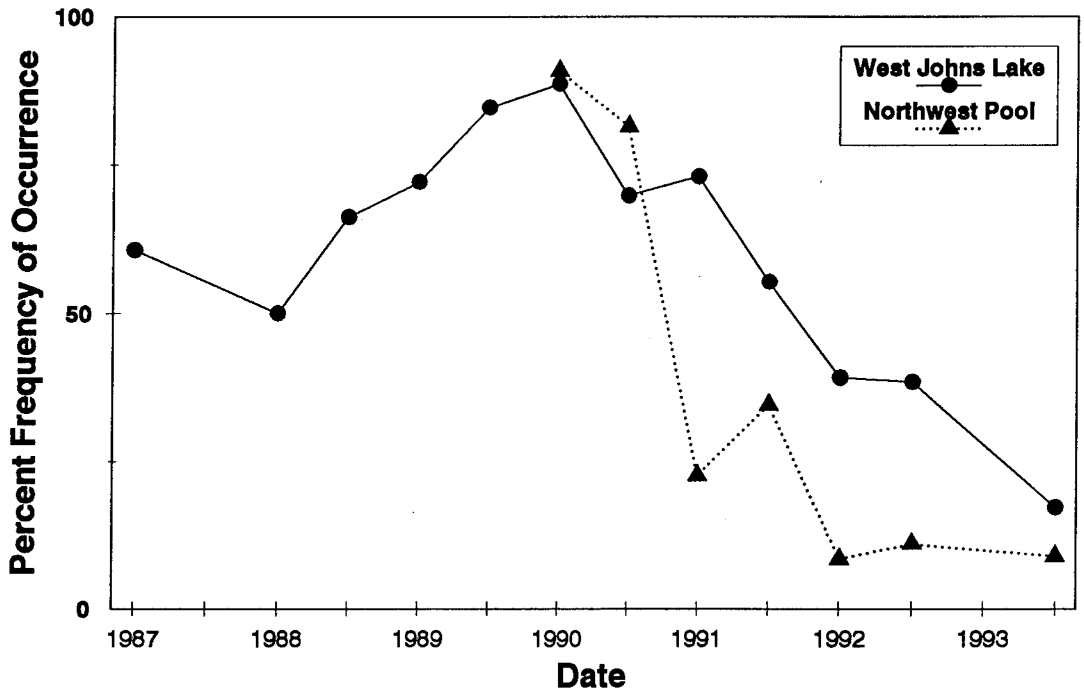

Figure 2. Percent frequency of occurrence of submersed macrophytes in west Johns Lake by year as determined by permanent recording fathometer transects

[1] Personal Communication, 1994, Dean G. Barber.

Table 3
Approximate Aerial Coverage in Hectares of Major Macrophyte Species in Johns Lake by Year[1]

Macrophyte Species	1986	1987	1988	1989	1990	1991	1992	1993
Hydrilla verticillata	7	166	122	132	24	0	0	trace
Nitella spp.	1	N/A	6	N/A	304	N/A	81	N/A
Eleocharis spp.	1	N/A	49	N/A	28	N/A	16	N/A
Utricularia spp.	24	N/A	49	N/A	101	N/A	223	N/A
Najas guadalupensis	2	N/A	24	N/A	57	N/A	32	N/A
Micranthemum glomeratum	20	N/A	6	N/A	7	N/A	7	N/A
Chara spp.	trace	N/A	trace	N/A	3	N/A	1	N/A
Ludwigia arcuata	49	N/A	32	N/A	36	N/A	36	N/A
Ceratophyllum demersum	2	N/A	30	N/A	65	N/A	32	N/A
Filamentous algae	7	N/A	4	N/A	trace	N/A	1	N/A
Panicum hemitomon	34	N/A	51	N/A	43	N/A	1	N/A
Panicum repens	24	N/A	34	N/A	44	N/A	60	N/A
Nymphaea odorata	7	N/A	26	N/A	40	N/A	49	N/A
Nuphar luteum	6	N/A	18	N/A	12	N/A	7	N/A
Fuirena scirpoidea	28	N/A	28	N/A	12	N/A	1	N/A
Typha spp.	97	N/A	122	N/A	105	N/A	81	N/A
Pistia stratiotes	0	0	10	13	4	4	1	3
Eichhornia crassipes	36	9	14	11	1	8	9	3
Total Macrophyte Coverage	450	600	700	800	910	800	748	650

[1] Data from Department of Natural Resources annual surveys.

reduction can be attributed to feeding by triploid grass carp after the hydrilla decline and appears consistent with observations by Trent et al. (1992) of similar feeding on palatable emergent species during late summer/early fall after individual triploid grass carp have achieved weights greater than 9 kg. This phenomenon may be related to a periodic need for cellulose in the diet (Das and Tripathi 1991) of larger triploid grass carp. Bonar et al. (1990) found a negative correlation between cellulose and macrophyte preference of triploid grass carp and probably is the more general characteristic of their food selection behavior.

Although native macrophytes currently exist at much lower levels than desired, the percent frequency of occurrence is expected to show increasing trends again in 2 to 5 years, with percent volume infestation of the water column increasing at a somewhat slower rate. These predictions are based on adequate water clarity (Figures 3 and 4) for macrophytes to reestablish in over 70 percent of the lake, longevity of triploid grass carp being only 10 years or less (Sutton and Vandiver 1986; Trent et al. 1992), and suspected decreased feeding rates of very large triploid grass carp (Osborne and Sassic 1981). Removal of a portion of the remaining triploid grass carp population in Johns Lake would accelerate regrowth of native plants, but without hydrilla as a habitat component if the hydrilla tuber reserves have indeed been depleted. A residual population of triploid grass carp in Johns Lake would still be effective for targeting any hydrilla regrowth that might occur, as has been observed over a longer term in Lake Fairview, Florida (Trent et al. 1992). Clapp et al. (1993) found that triploid grass carp in Lake Yale, Florida, preferred hydrilla over native macrophytes, feeding on hydrilla disproportionately to its abundance in the lake.

Comparison of management techniques

Actual dollar expenditures for materials used in this integrated hydrilla management program compared with the projected expenditures for materials in a strictly herbicidal hydrilla management program using fluridone (Table 4) results in a cost savings of over $200,000 in the time span of 9 years (1986-1994) by using the integrated approach. This disparity in management program materials costs will continue to grow, as little or no

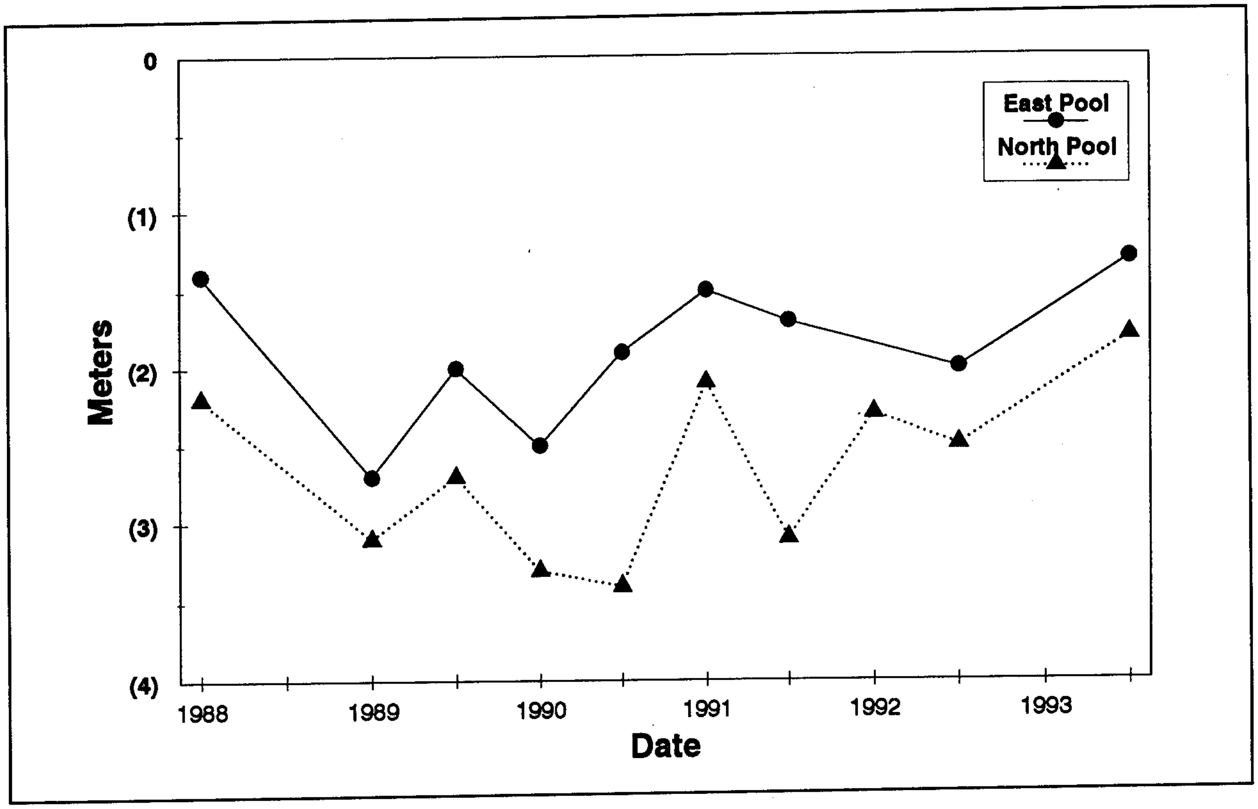

Figure 3. Secchi visibility readings in west Johns Lake by year

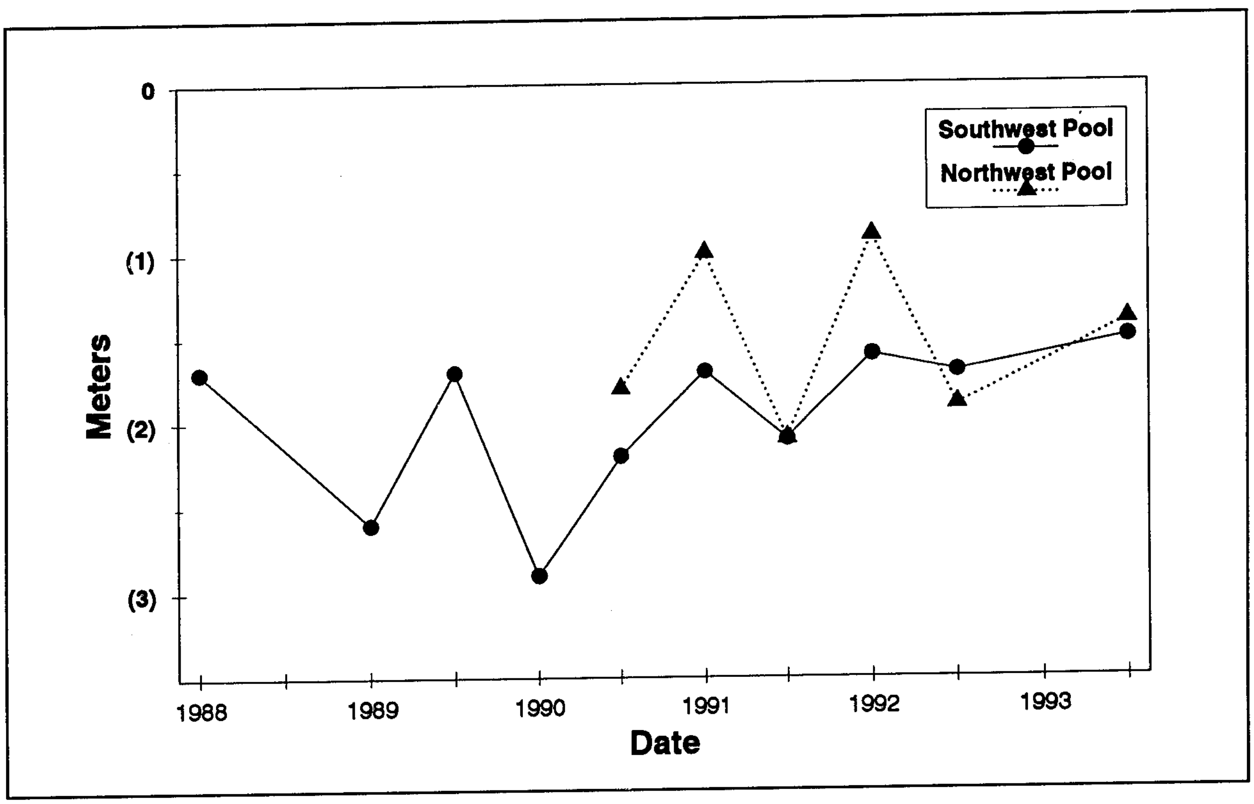

Figure 4. Secchi visibility readings in west Johns Lake by year

Table 4
Comparison of Materials Cost for Integrated Hydrilla Management in Johns Lake Versus Projected Costs for a Strictly Herbicidal Hydrilla Management Program Using Fluridone

	Integrated		
	Fish[1]	Contact Herbicides[2]	Fluridone[3]
1986		$608	$608
1987		$13,376	$13,376
1988	$18,240	$22,254	$100,000
1989	$13,280	$28,080	
1990	$5,012		
1991			$100,000
1992			
1993			
1994			$100,000
Total	$36,532	$64,318	$313,984
Total Integrated	$100,850		

[1] Actual expenditures for triploid grass carp stocked for hydrilla management in Johns Lake.
[2] Actual expenditures for contact herbicides used for hydrilla management in Johns Lake.
[3] Projected herbicide costs for fluridone based on conservative estimates of an 8.3-percent (81.2-ha) lake treatment at a rate of 4.7 L fluridone/hectare treated, cost of $1,000/3.8 L of fluridone, and repeated whole-lake fluridone treatments being required every third year in Johns Lake. The 1986 and 1987 figures are actual costs for contact herbicides, as the contact herbicides would have been utilized preliminary to the whole-lake fluridone treatments.

expenditures for future hydrilla control are expected in the current integrated program because of the suspected depletion of tuber reserves in the substrate and the residual population of triploid grass carp continuing to target any hydrilla that might regrow.

Although significant materials cost savings were realized with this integrated program, there are many other management considerations (Table 5). It is readily apparent that there are numerous advantages and disadvantages to both management techniques. Therefore, selection of a management technique depends heavily on established objectives, and materials cost may or may not be of prime consideration. The integrated approach relies more heavily on effective monitoring in order to plan ongoing herbicide treatments and triploid grass carp stockings (or removals). Timing and degree of herbicide treatments and triploid grass carp stockings are usually critical to the effectiveness and ultimate success of the integrated approach. However, proper timing is difficult to achieve (Sutton et al. 1986) because of unavailability of triploid grass carp, delays in funding for the purchase of herbicides, and many other factors. As

Table 5
Comparisons of Advantages and Disadvantages of Integrated Hydrilla Management and Hydrilla Management Using Strictly Herbicides (fluridone)[1]

Advantages	Disadvantages
Integrated	
* Fewer grass carp and less quantity and frequency of herbicide use than with either method used alone. * Significant reduction or sometimes elimination of the need for future and long-term hydrilla control. * Possible elimination of hydrilla from the water body. * Significant long-run cost savings. * Multiple uses of the water body maintained. * Extended benefits from contact herbicide treatments seen where more than 1 ha of hydrilla control is achieved for every hectare treated.	* Nontarget effects - temporary loss of habitat. * Proper timing and degree of integration is difficult to achieve. * Frequent monitoring required. * Removal of grass carp may be required, adding additional cost and effort. * Mortality of the triploid grass carp is not known, adding uncertainty to estimating long-term stocking rates and the need for ongoing hydrilla control operations.
Fluridone	
* Normally good selectivity for hydrilla - good chance to maintain native macrophytes. * Ease of application. * Minimal monitoring required. * Future technology to eliminate hydrilla from a water body?	* High cost. * High solubility - difficult to obtain partial control. * Regrowth of hydrilla occurs, and continual and costly retreatment necessary with current technology. * Some native plant species such as water lilies can be reduced or eliminated. * Some lake uses periodically impaired.

[1] This is not an exhaustive list, as there are many other considerations depending on the water body, environmental conditions, management objectives, etc.

seen in this example, the integrated approach can result in significant nontarget effects when hydrilla is either the dominant macrophyte or is increasing exponentially. A fluridone hydrilla management program in Johns Lake would have been more selective for hydrilla and most of the offshore maidencane community probably would have been maintained, but nontarget effects from fluridone can also occur and may be unavoidable in some cases (Farone and McNabb 1993). For example, fragrant water lily in Johns Lake could have been significantly reduced because of nontarget effects from fluridone (Farone and McNabb 1993), but was one of the species that increased in this integrated program. Monitoring programs to identify new infestations of hydrilla are extremely important. Much more effective results can be achieved in any hydrilla management program if hydrilla control measures are initiated before hydrilla begins increasing exponentially, as regrowth of hydrilla can only occur in rather limited areas because of the fact that a large tuber reserve in the substrate does not exist. Also, there is much less chance for nontarget effects when hydrilla populations are not allowed to expand.

The power of the integrated approach can be seen by evaluating results from the contact herbicide treatments. Johns Lake endothall treatments in 1988 resulted in 1 to 2 years of hydrilla control before regrowth was observed in the treatment plots. Without integration, only 2 to 6 months control is achieved with contact herbicides. With integration, control outside the endothall treatment plots were noted in Johns Lake, resulting in 0.6 to 0.8 ha of hydrilla controlled for every 0.4 ha treated (1.5 to 2:1 ratio). Similar results were seen in a smaller lake (unpublished data) where integrated endothall treatments (20.6 ha) resulted in 60.7 ha of hydrilla control (3:1 ratio). Without integration, the control to treatment ratio using contact herbicides usually is 1:1.

Largemouth bass response to habitat alteration

Canfield and Hoyer (1992) cited conflicting relationships between macrophyte abundance and largemouth bass abundance. Bettoli et al. (1992) related minimal macrophyte coverage with early piscivory and faster growth rates of largemouth bass. However, moderate to dense macrophyte coverage appears to be especially important for largemouth bass recruitment in large numbers and formation of quality year-class strength (Wiley et al. 1984; Durocher, Provine, Kraai 1984; Moxley and Langford 1985; Porak et al. 1990). Therefore, it is reasonable to expect that a significant reduction in macrophyte habitat would lead to significant changes in largemouth bass recruitment, abundance, food habits, and growth. Background data for largemouth bass populations in Johns Lake were not available, but electroshocking surveys were initiated in fall 1990 to obtain current data. Electroshocking has some severe limitations (Hardin and Conner 1992), especially for sampling open-water largemouth bass populations. However, electroshocking data should show some trends, at least for largemouth bass in a portion of Johns Lake.

Electrofishing CPUE for largemouth bass (Table 6) decreased from 1990 until fall 1993 when some recovery in CPUE was noted. Preliminary data from spring 1994 indicate the highest CPUE values for some largemouth bass size classes since sampling was initiated. Stock density indices for largemouth bass (Table 7) show PSD values within or near the preferred range (40 to 70) throughout the sampling period. Similarly, largemouth bass RSD-14 values were within or near the preferred range (10 to 40), but RSD-15 values were somewhat below the preferred range (10 to 40), especially during the fall sampling periods. The largemouth bass RSD-20 values were within the preferred range (1 to10). Relative weight values (W_r) from spring electrofishing surveys of largemouth bass (Table 8) were within or above the preferred range (95 to 105) for small fish (<20.3 cm), while W_r values for larger fish (>20.3 cm) were below the preferred range in spring 1991 and 1992, but improved during spring 1993. In fact, highest W_r values for all largemouth bass size classes were observed during spring 1993. Tournament angler catch data from fall 1993

Table 6
CPUE (number/minute) for Various Largemouth Bass Length Groups Collected from Johns Lake During Biannual Electrofishing Surveys 1990-1993[1]

Year	All Sizes	>20.3 cm	>30.5 cm	>35.6 cm	>38.1 cm	>50.8 cm
Fall 1990	1.6	0.7	0.3	0.1	0.1	—
Spring 1991	0.9	0.7	0.3	0.1	0.1	<0.1
Fall 1991	0.7	0.4	0.2	<0.1	<0.1	—
Spring 1992	0.7	0.5	0.2	0.1	<0.1	—
Fall 1992	0.6	0.4	0.2	<0.1	<0.1	<0.1
Spring 1993	0.8	0.4	0.2	<0.1	<0.1	—
Fall 1993	1.1	0.5	0.2	0.1	0.1	<0.1

[1] Unpublished data collected by Central Region Florida Game and Fresh Water Fish Commission Regional Fisheries project.

Table 7
Stock Density Indices (PSD and RSD) for Largemouth Bass Collected from Johns Lake During Biannual Electrofishing Surveys 1990-1993[1]

Year	PSD	RSD-14	RSD-15	RSD-20	RSD-24
Fall 1990	42	15	7	—	—
Spring 1991	61	20	13	4	—
Fall 1991	38	5	2	—	—
Spring 1992	38	13	4	—	—
Fall 1992	50	10	5	3	—
Spring 1993	42	10	10	—	—
Fall 1993	52	23	12	2	—

[1] Unpublished data collected by Central Region Florida Game and Fresh Water Fish Commission Regional Fisheries project. Objective range for PSD values is 40 to 70, 10 to 40 for RSD-14/RSD-15, and 1 to 10 for RSD-20.

Table 8
Mean Relative Weight Values (W_r) for Largemouth Bass Length Groups Collected from Johns Lake During Spring Electrofishing Surveys 1991-1993[1]

Year	<20.3 cm	20.3 to 27.9 cm	30.5 to 35.6 cm	38.1 to 48.3 cm	>50.8 cm
1991	103	86	85	85	95
1992	101	91	85	86	—
1993	118	101	91	95	—

[1] Unpublished data collected by Central Region Florida Game and Fresh Water Fish Commission Regional Fisheries project. Objective range for W_r values is 95 to 105.

(Table 9) show that the harvestable largemouth bass catch rates (0.17 fish/hour) were slightly below the statewide average of 0.25 fish/hour. CPUE for all size classes was 1.00 fish/hour. Mean weight of harvestable largemouth bass weighed in was 0.8 kg, and large fish weighed 3.8 kg.

These data indicate that largemouth bass in Johns Lake may be adapting to a changed forage base because of habitat alteration. During 1990, largemouth bass food habits probably were more closely related to a macrophyte-oriented forage base. By 1993, largemouth bass food habits appear to be more closely related to an open-water forage base (primarily shad, *Dorosoma petenense* and *D. cepedianum*). In fact, increased frequency of largemouth bass schooling behavior was noted during vegetation sampling trips in 1992 and 1993. However, recruitment may be depressed because of inadequate habitat, and the suspected open-water population of largemouth bass may not be able to withstand consistent harvest.[1] If

1 Personal Communication, 1994, William Coleman.

Table 9
Angler Catch Data for Largemouth Bass Caught During a Mini-Bass Small Boat Tournament Trail Event During November 1993 in Johns Lake, Florida[1]

CPUE all sizes	1.00/hr
CPUE harvestable fish (≥35.6 cm)	0.17/hr
CPUE nonharvestable fish (<35.6 cm)	0.83/hr
Mean weight of harvestable fish (≥35.6 cm)	0.8 kg
Big fish	3.8 kg

[1] Unpublished data collected by Central Region Florida Game and Fresh Water Fish Commission Regional Fisheries project.

the predicted increase in native macrophytes occurs over the next 2 to 5 years, then largemouth bass food habits should gradually shift again toward a macrophyte-oriented forage base and recruitment should improve as well.

Conclusions

Although moderate coverages of hydrilla have been indicated to aid largemouth bass recruitment and forage production, hydrilla management is extremely costly and fast becoming unaffordable in the State of Florida. Long-term economic and political conditions can be expected to be much worse at the State level (Stocker 1993) and also nationally and internationally (Burkett 1991; Robertson 1991). Oligotrophic to moderately eutrophic lakes generally should be able to support adequate stands of native macrophytes valuable as largemouth bass habitat, without the need for the exotic macrophyte hydrilla as a habitat component and the associated expense for hydrilla management. This case study has indicated the possibility of using integrated hydrilla control to eliminate hydrilla from an individual water body, and this result was previously thought to be unattainable. In the long term, temporary loss of habitat, temporary decreases in largemouth bass recruitment rates, possible elimination of hydrilla, and significant cost savings incurred with integrated hydrilla management may be a more desirable management result than continual and costly treatments with herbicides alone. Charyev (1984) made similar observations, in that despite some negative effects from grass carp, the Kara Kum Canals in Turkmenistan were in much better condition than if grass carp had not been utilized as a management tool. Langeland (1990) stated that in most cases, the detrimental effects of hydrilla infestations far outweigh the potential benefits. However, management objectives must be clearly defined for an individual water body before accepting such trade-offs.

Acknowledgments

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Integrated Hydrilla Management Plan Utilizing Herbicides and Triploid Grass Carp in Lake Istokpoga

by
David Eggeman[1]

Introduction

Hydrilla (*Hydrilla verticillata*) was first observed in Lake Istokpoga in 1979. By 1988, it had expanded to an estimated 8,092 ha. Herbicide (Fluridone) treatments began in 1987 and continued each spring through 1991 (Table 1). Although herbicide treatments successfully reduced hydrilla coverage on Lake Istokpoga, their effectiveness was short-lived. Annual vegetation surveys conducted by the Department of Environmental Protection (DEP) in October showed that hydrilla regrew each year (Table 1). This rapid expansion of hydrilla in Lake Istokpoga after spending approximately $2.3 million for hydrilla control caused the DEP, Florida Game and Fresh Water Fish Commission (GFC), U.S. Army Corps of Engineers, and Highlands County to re-evaluate hydrilla management options on this lake. A jointly coordinated interagency project to integrate herbicides and triploid grass carp (*Ctenopharyngodon idella*) was developed in 1991 and implemented in 1992. The project goal is to prolong the effectiveness of a large-scale herbicide treatment by maintaining hydrilla surface coverage below 2,832 ha (7,000 acres). The initial stocking level of 125,000 triploid grass carp was determined based on hydrilla coverage after the 1992 large-scale herbicide treatment (Table 2).

Table 1
Cost of Hydrilla Treatments and Hydrilla Coverage by Year in Lake Istokpoga

Year	Cost, $ (Jan-Mar)	Hydrilla, ha (Jan)	Hydrilla, ha (Oct)
1988	153,000	1,450	5,350
1989	1,218,000	5,350	4,300
1990	467,000	4,300	3,500
1991	450,000	3,500	4,300
1992	1,200,000	4,300	800
1993	656,000[a]	800	3,650
1994	0	3,650	?

[a] Triploid grass carp.

Table 2
Triploid Grass Carp Stocking Options for Lake Istokpoga

Number Proposed	Criteria
75,000	0 ha Hydrilla
125,000	<2,023 ha Hydrilla
125,000	2,023-2,832 ha Hydrilla[1]
0	>2,832 ha Hydrilla

[1] Additional herbicide treatment required.

A potential future additional stocking of 25,000 triploid grass carp will be based on percent coverage of hydrilla regrowth (Table 3).

Table 3
Supplemental Triploid Grass Carp Stocking Options for Lake Istokpoga

Option	% Occurrence of Hydrilla[1]	Action
1	0 to 15 %	No Stocking
2	15 to 35%	Supplemental stocking at 2.5 fish/ha (25,000 fish)
3	more than 35%	No stocking[2]

[1] Percent occurrence lakewide.
[2] Additional stocking at low rates would not be effective.

Study Site

Lake Istokpoga is approximately 11,328 ha (28,000 acres) located in Highlands County in south-central Florida. The lake has two inlets (Josephine and Arbuckle creeks) and two outlets (Istokpoga and C-41a canals). Josephine Creek has a fixed structure to prevent fish

[1] Florida Game and Fresh Water Fish Commission, Tallahassee, FL.

movement. Arbuckle Creek is navigable and connects Lakes Istokpoga and Arbuckle. The two outflow canals connect to the Kissimmee River and Lake Okeechobee.

Methods

Hydrilla was treated with 2,177 kg (4,800 lb) active ingredient fluridone in January-March 1992. Triploid grass carp (125,000; 30-cm minimum) were stocked in October 1992 through March 1993. Aquatic vegetation has been and will continue to be mapped each October using a laptop computer and Loran-C to provide locations and acreage. Quarterly transects will be run using a recording depth finder to determine hydrilla regrowth. Largemouth bass (*Micropterus salmoides*) and total fish populations will be monitored by semi-annual electrofishing (12 hr pedal time) and annual block-nets (six 0.4-ha). Water quality will be sampled and analyzed quarterly. Six enclosures to exclude triploid grass carp were placed in submerged vegetation to serve as control sites so that dynamics of vegetation not exposed to triploid grass carp feeding could be observed. Creel surveys will be conducted during peak fishing season (November-March) to determine angler effort and catch rates.

Results

Hydrilla was reduced by herbicides to approximately 800 ha by October 1992, of which most was surfaced out. As a result, 125,000 triploid grass carp were stocked between October 1992 and March 1993. Hydrilla expanded to approximately 3,650 ha by February 1994; however, less than 40 ha were surfaced out.

Discussion

Fish population and water quality data were collected prior to triploid grass carp stocking. Vegetation mapping has been done each October. It is too early to draw any conclusions on effects of triploid grass carp stocking on hydrilla coverage, water quality, fish populations, or angler creel. Data will continue to be collected until an October vegetation survey reveals more than 2,832 ha of surfaced out hydrilla.

Texas and South Carolina Case Histories

Lake Conroe Fisheries - Population Trends Following Macrophyte Removal

by
Mark A. Webb,[1] *Jeffrey C. Henson,*[1] *and Michael S. Reed*[2]

Introduction

Lake Conroe (located near the town of Conroe, TX) was impounded in 1973 by the San Jacinto River Authority and the City of Houston for water supply. The 21,000-acre reservoir is somewhat clear with Secchi disk readings of 2 to 4 ft. The lower two-thirds of the reservoir is heavily developed with approximately the upper one-third of the reservoir in national forest.

Pressure from lake area landowners and developers demanding relief from the heavy infestation of macrophytes (primarily hydrilla (*Hydrilla verticillata*)) resulted in the stocking of 270,000 diploid grass carp (*Ctenopharyngodon idella*) into Lake Conroe in the early 1980s. The effects of the resulting loss of macrophytes on the fish populations of Lake Conroe were evaluated and documented by Texas A&M University. The Texas A&M project report (Klussmann et al. 1988) noted that by 1983 macrophytes had been almost completely removed from the reservoir with a resulting increase in primary productivity. However, by the conclusion of the investigation in 1986, most nutrients had returned to pretreatment levels. Notable changes occurring in the fish community included overall decreases in sunfishes (*Lepomis* spp.), crappies (*Pomoxis* spp.), and intermediate size largemouth bass (*Micropterus salmoides*) and overall increases in threadfin shad (*Dorosoma pretense*), yellow bass (*Morone mississippiensis*), white bass (*M. chrysops*), and channel catfish (*Ictalurus punctatus*). These changes were attributed to an increase in pelagic habitat and primary productivity and a decrease in littoral habitat. The focus of this paper is on the status of the Lake Conroe fisheries since the termination of the Texas A&M project in 1986.

Methods

Fishes were collected by electrofishing (2 hr at eight stations), gill netting (15 net nights at 15 stations), and frame netting (15 net nights at 15 stations) unless otherwise noted (Figure 1). Catch per unit effort (CPUE) was recorded for electrofishing as the number of fish caught per hour of actual electrofishing and for gill and frame nets as the number of fish caught in one net set overnight. Sampling statistics (CPUE for various length categories) and structural indices (proportional stock density (PSD), relative stock density (RSD), and relative weight (Wr)), were calculated for target fishes according to Anderson and Gutreuter (1983) and Childress (1989). An aquatic vegetation survey was conducted

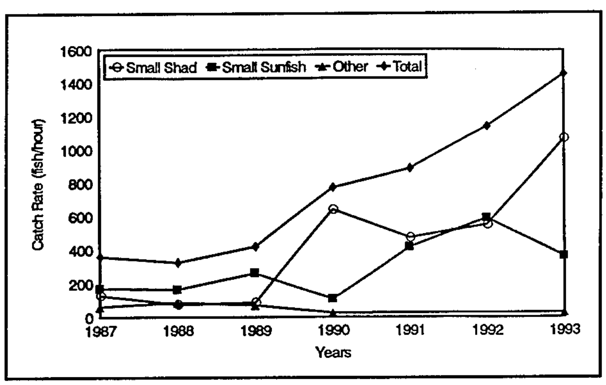


Figure 1. Catch per hour (CPUE) of small forage fishes (≤4 in.) from electrofishing, Lake Conroe, Texas, fall 1987-1993

[1] Texas Parks and Wildlife Department, Inland Fisheries, Bryan, TX.
[2] Texas Parks and Wildlife Department, Inland Fisheries, Mathis, TX.

in accordance with procedures in Texas Parks and Wildlife Department (1993). Primary productivity data were collected approximating methods given in Klussmann et al. (1988).

Results

Aquatic macrophytes

Aquatic macrophyte density remains insignificant, with only 8.3 acres of floating native vegetation found during a 1993 vegetation survey.

Forage fishes

Forage was dominated by threadfin shad, small gizzard shad (*D. cepedianum*), longear sunfish (*L. megalotis*), and small bluegill (*L. macrochirus*). During 1987-1993, relative abundance in electrofishing increased for all four species (Figure 1), with the most notable increases in threadfin shad from a low of 50.5 per electrofishing hour in 1989 to a high of 1,050.0 per electrofishing hour in 1993. Increased forage production in Lake Conroe is likely due to elevated primary productivity as indicated by 1990 nitrate nitrogen (0.59 mg/L) and total nitrogen concentrations (0.87 mg/L) that were well above values recorded by Texas A&M during 1980-1986 (Klussmann et al. 1988). Increased nitrogen levels could be due to increased development around Lake Conroe's shore leading to increased fertilizer runoff and lateral line seepage.

Largemouth bass

Electrofishing catch rates suggest a decrease in the relative abundance of largemouth bass over time, with catch per unit effort from electrofishing dropping from a high of 202 per hour in 1989 to a low of 53 per hour in 1993 (Figure 2). Although population structure indices (PSD and RSD-14) have shown some increase over time, catch rates of ≥14 in. fish have remained similar, indicating that changes in structural indices are primarily due to a decrease in stock-size subquality fish. The minimum length limit for largemouth bass at Lake Conroe was increased from 14 to 16 in. in 1993 to increase average size of fish caught and fish harvested and to increase overall bag weights for consumption and tournament weigh-ins.

Channel catfish

Lake Conroe supports a high density, high quality channel catfish fishery that is still expanding. Despite removal of 0.5-, 3.5-, and 4.0-in. mesh panels from standard gill nets in 1992, relative abundance estimates of channel catfish have continued to increase (Figure 3). Expansion of the channel catfish population is likely due to increased open-water benthic habitat after macrophytes were removed coupled with increased primary productivity. The minimum length limit for channel catfish at Lake Conroe was raised from 9 to 14 in. in 1992 in order to increase the average size of fish caught and fish harvested, to increase the overall yield to the angler, and to protect the channel catfish population to a level of approximately 50-percent maturity.

White crappie and black crappie

Both white crappie (*P. annularis*) and black crappie (*P. nigromaculatus*) are present in low densities in Lake Conroe. Following removal of macrophytes, catch rates of crappies from frame nets fell to very low levels (Figures 4 and 5). White crappie rebounded somewhat after subadult (approximately 4 in.) fish were stocked and a 10-in. minimum length limit was imposed during 1990.

White bass

The Lake Conroe white bass population is characterized by sporadic year class strength (Figure 6) affected by tributary inflows during spawning runs among other things. The white bass minimum length limit was increased from 10 to 12 in. in 1992 to protect fish to maturity, to increase the average size of fish caught and fish harvested, and to increase yield to the angler from sporadic year classes.

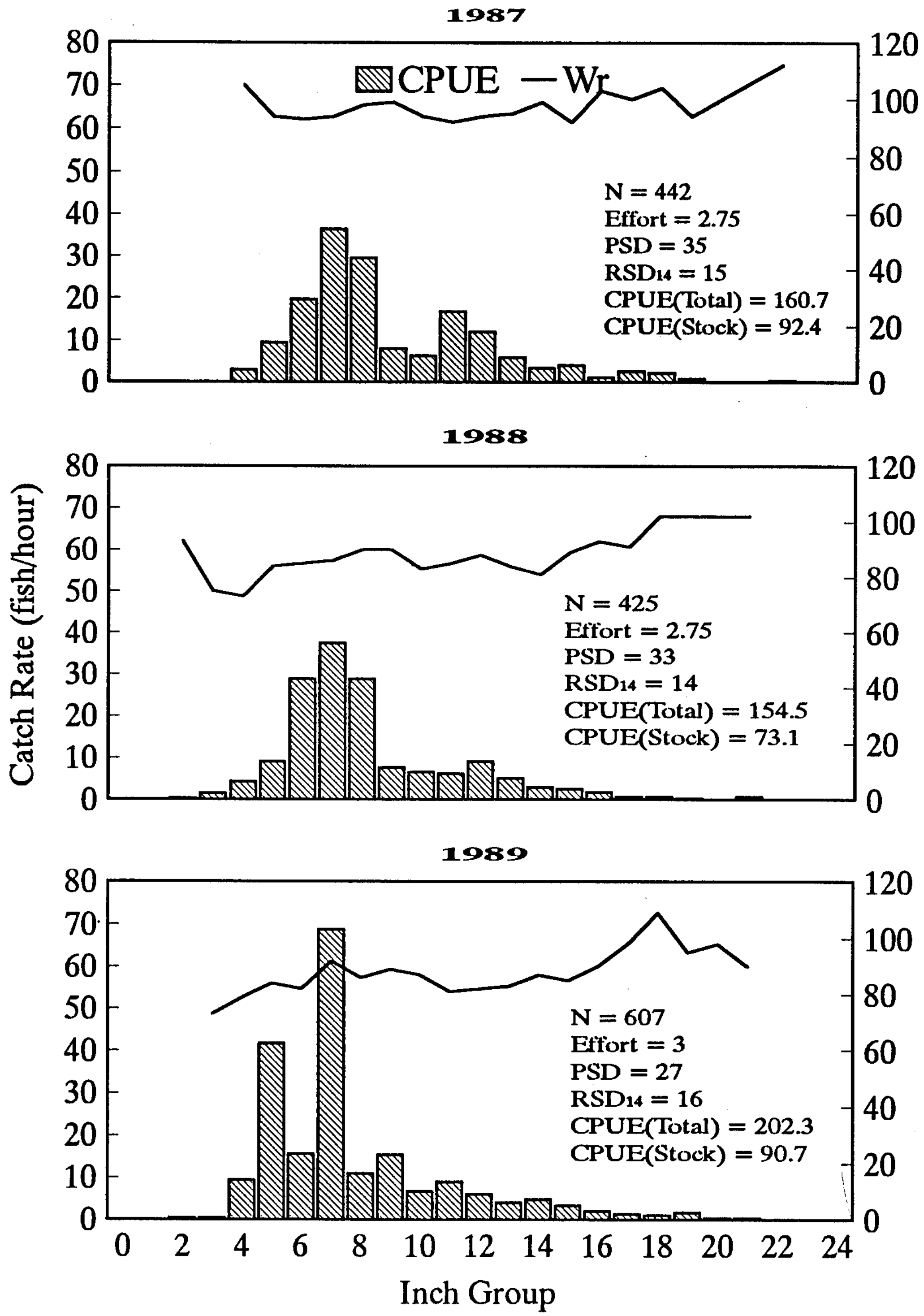
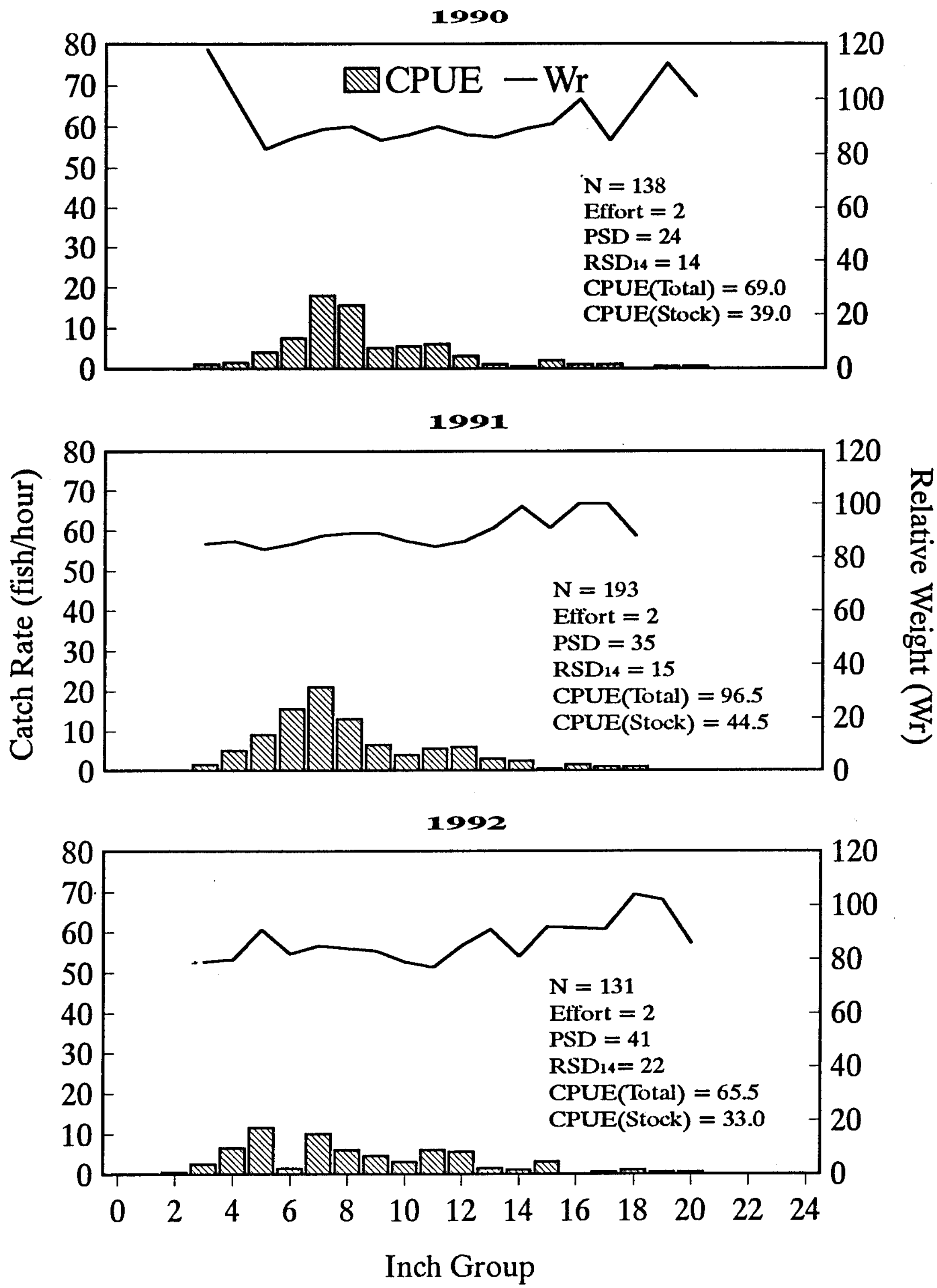


Figure 2. Catch per hour (CPUE) and relative weight (Wr) of largemouth bass collected by electrofishing, Lake Conroe, Texas, fall 1987-1993 (Sheet 1 of 3)



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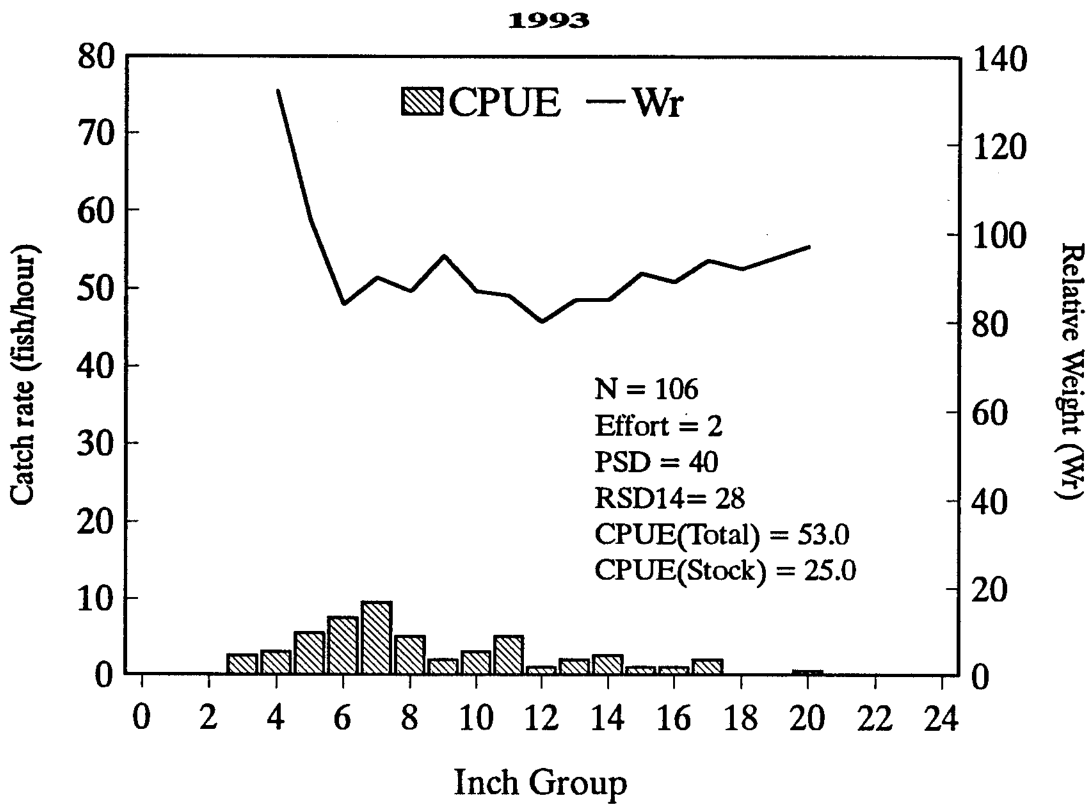

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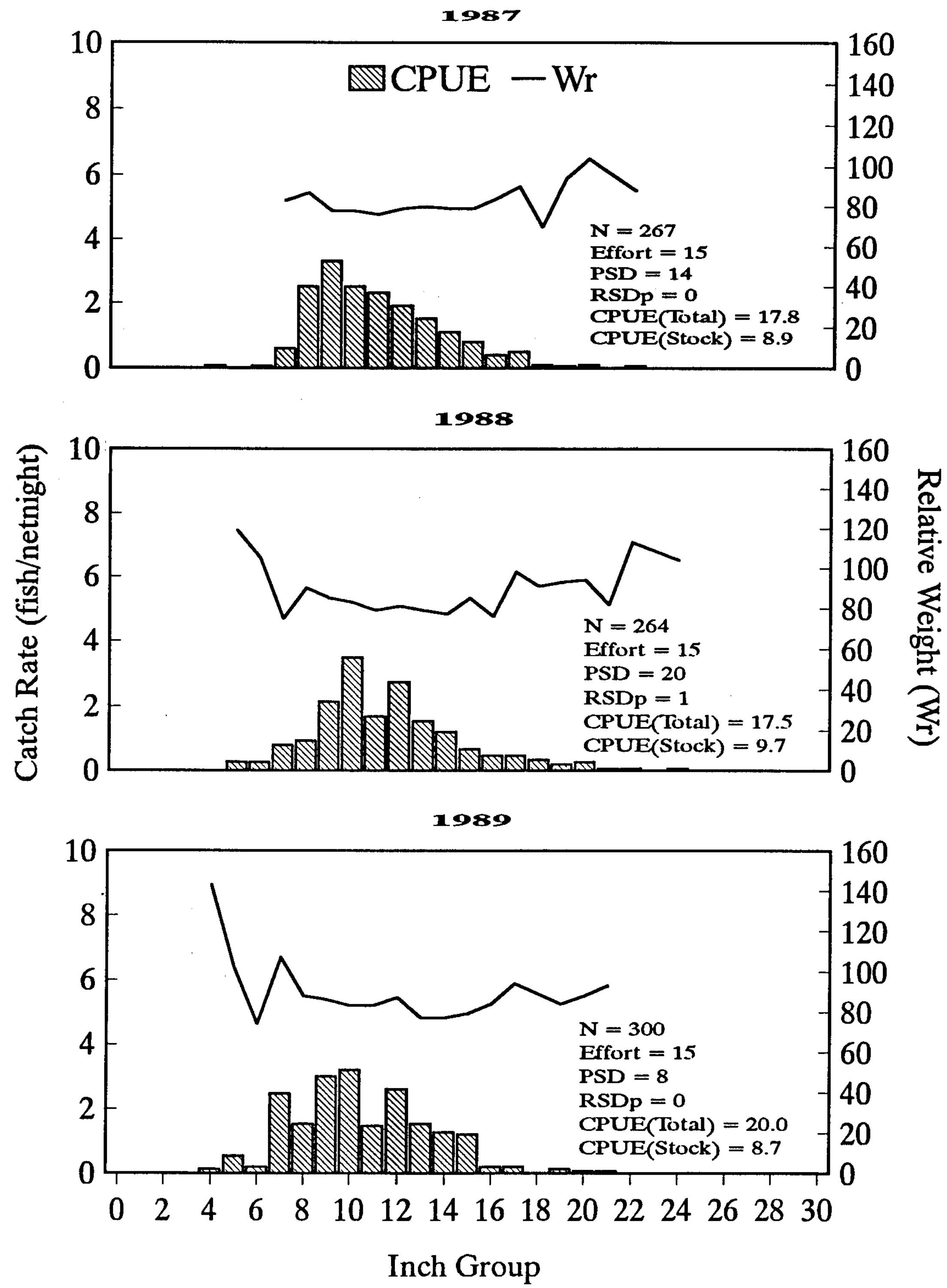
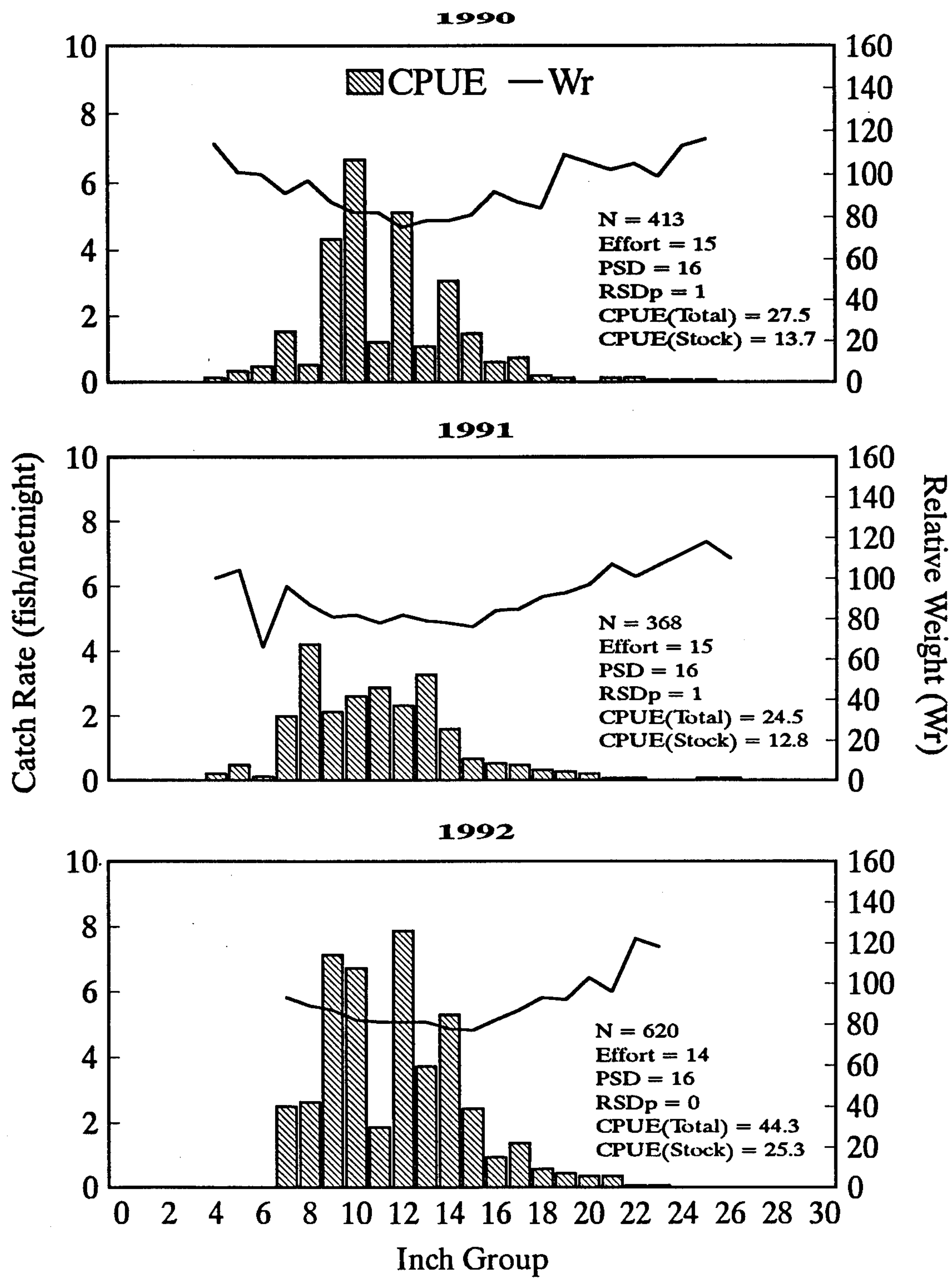


Figure 3. Catch per net night (CPUE) and relative weight (Wr) of channel catfish collected by gill netting, Lake Conroe, Texas, spring 1987-1993 (Sheet 1 of 3)



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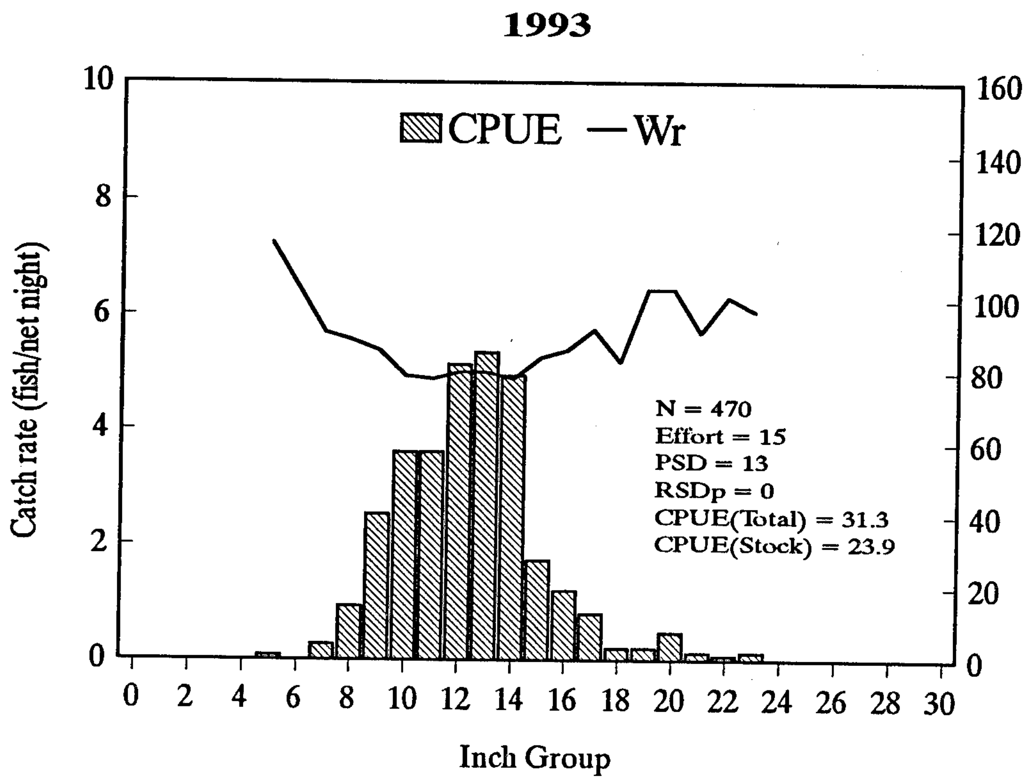

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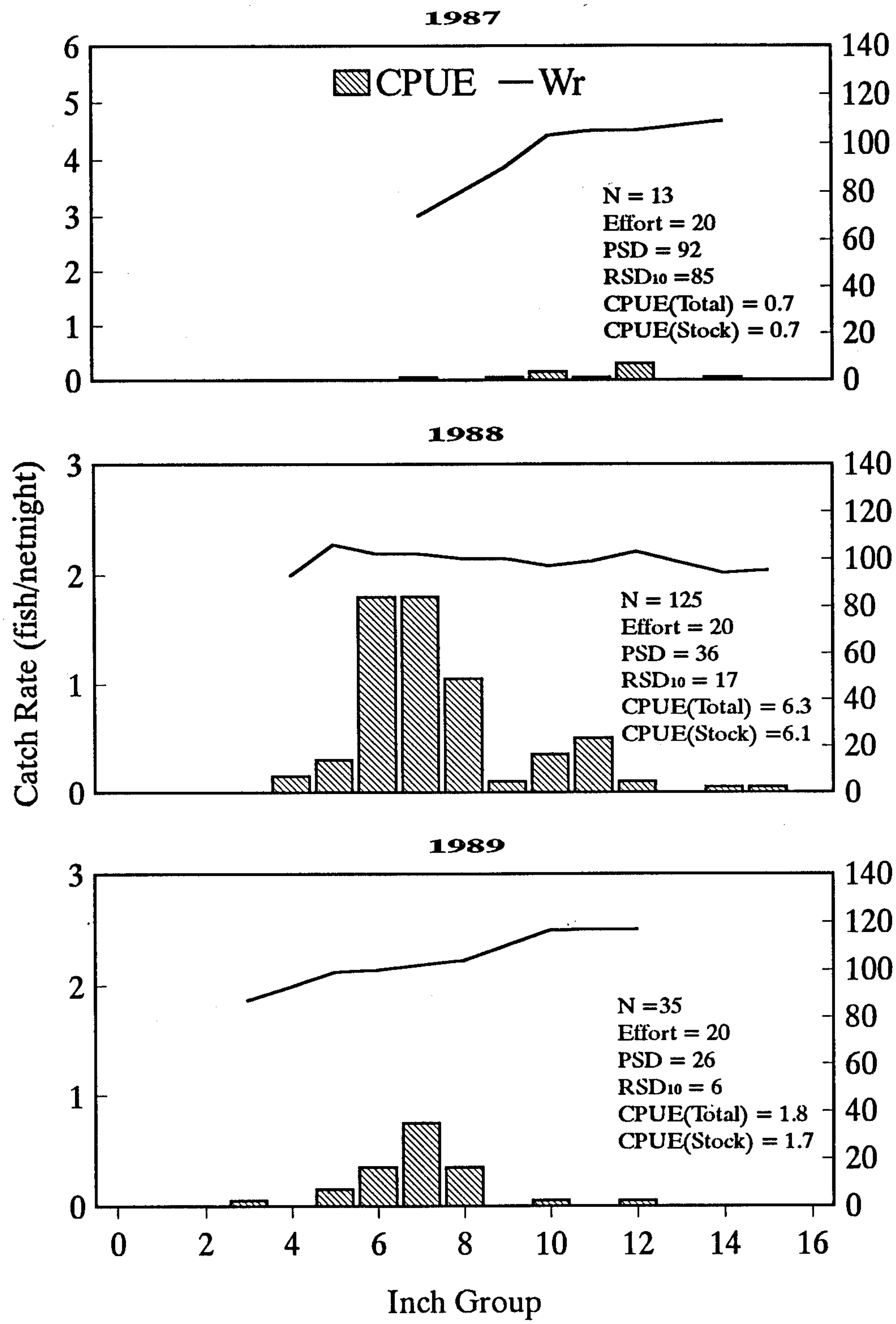
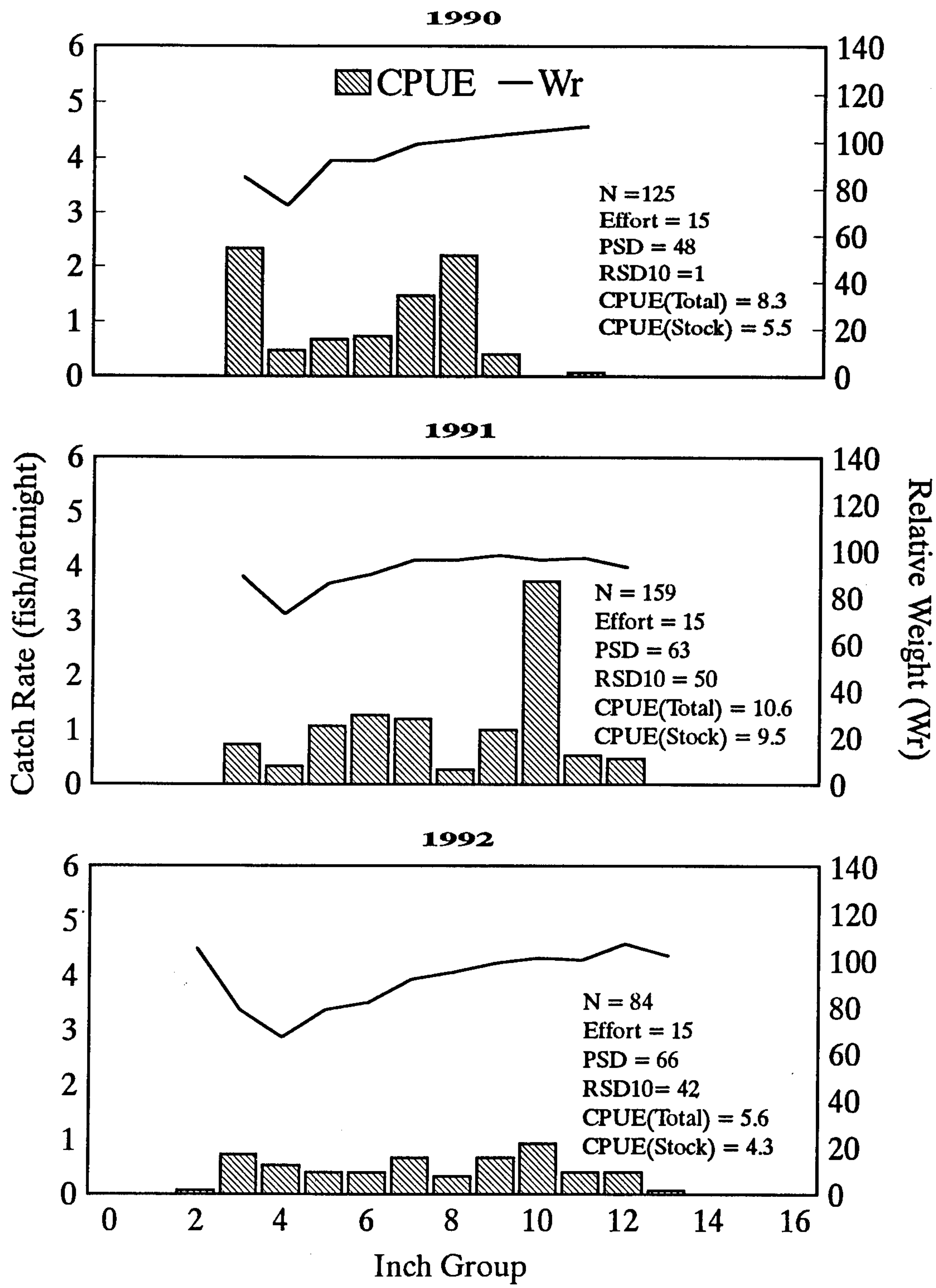


Figure 4. Catch per net night (CPUE) and relative weight (Wr) of white crappie collected by frame netting, Lake Conroe, Texas, fall 1987-1993 (Sheet 1 of 3)



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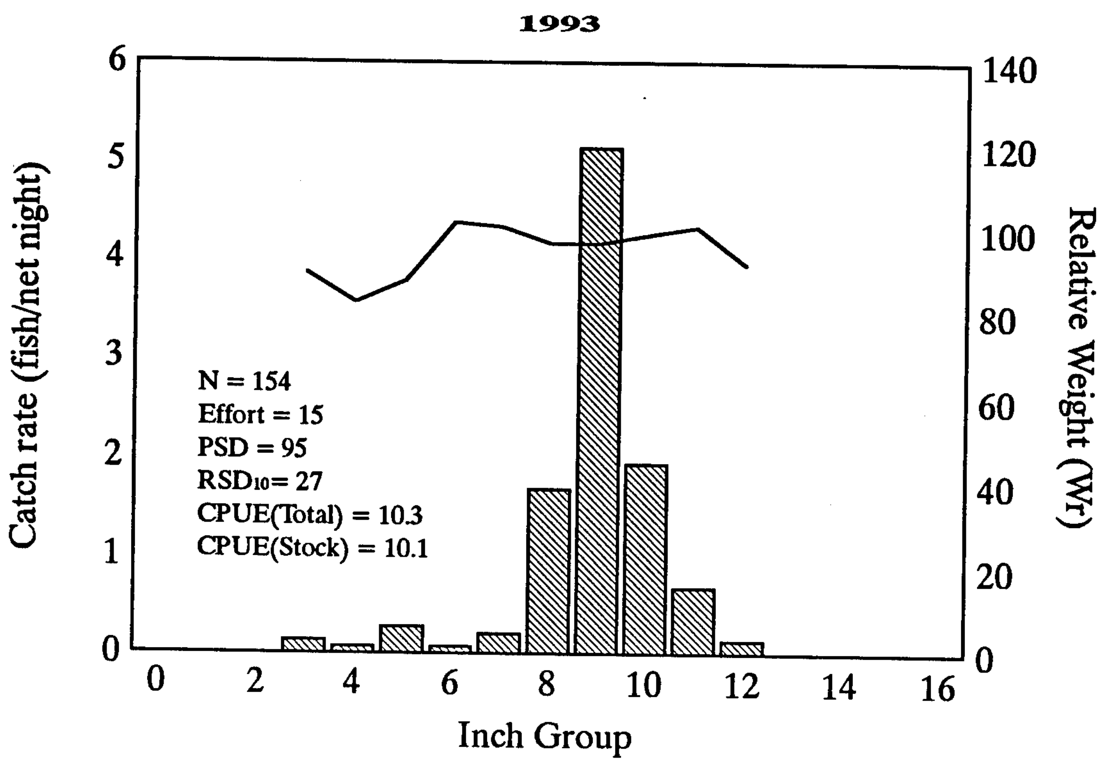

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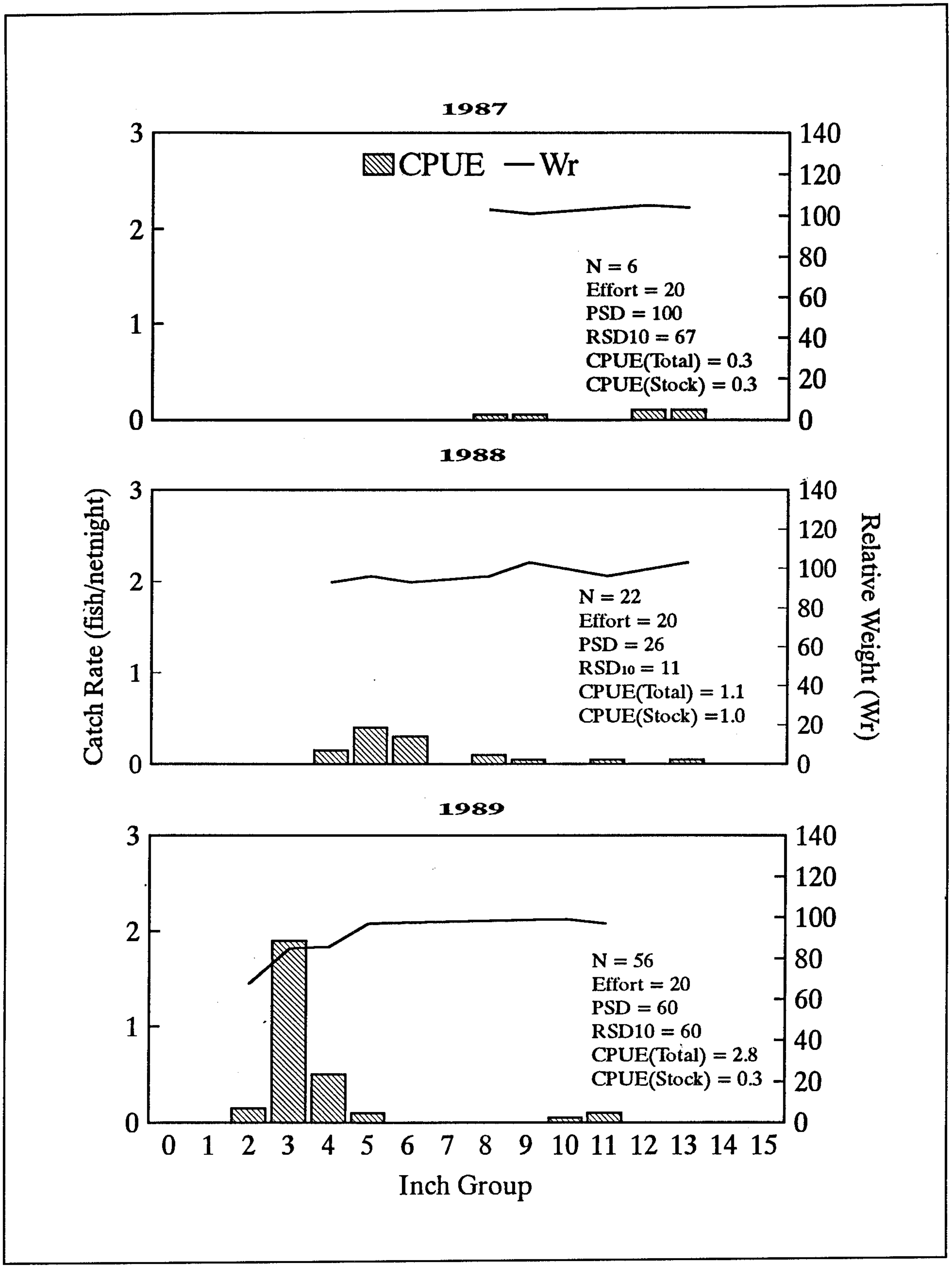
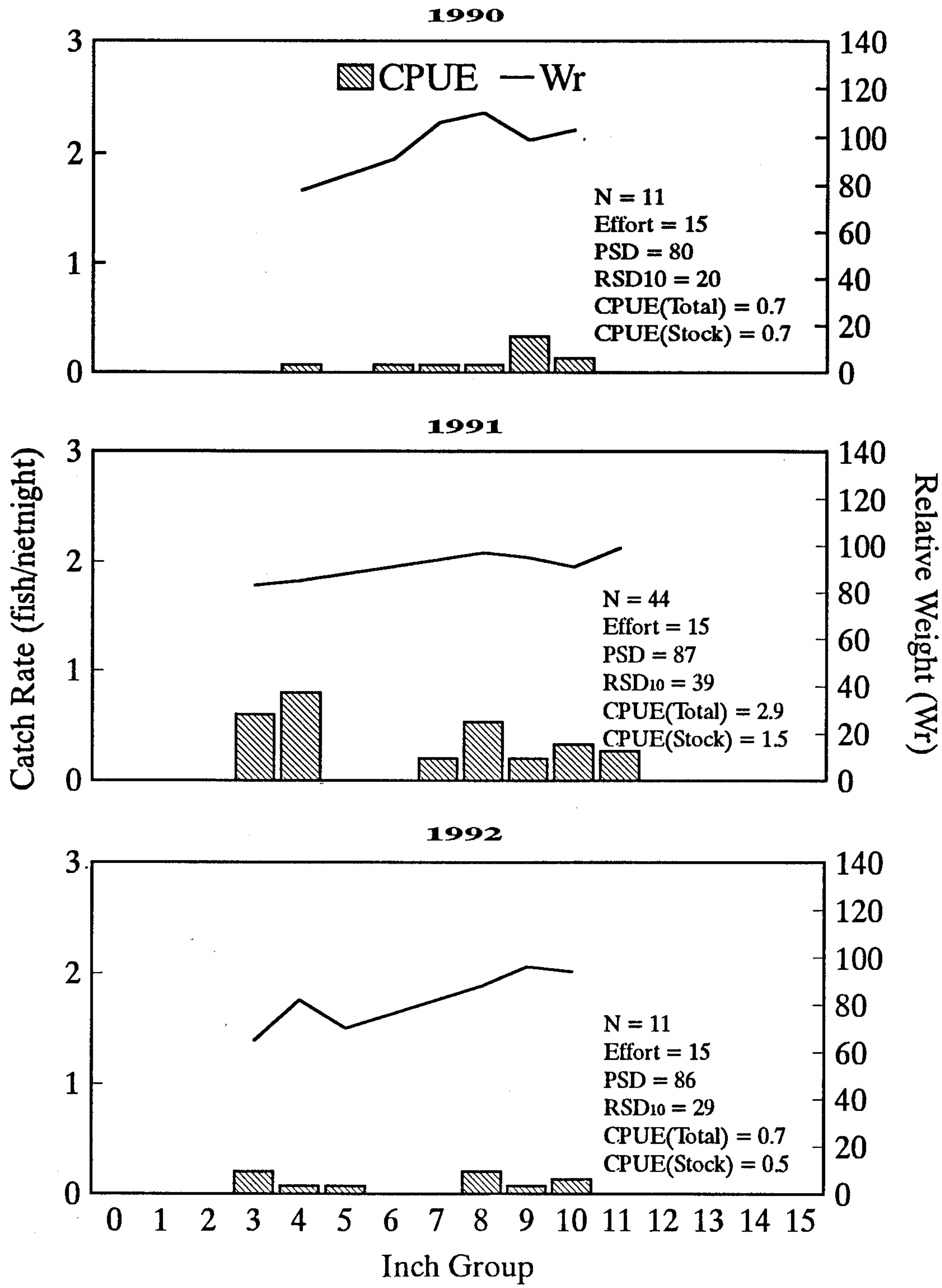


Figure 5. Catch per net night (CPUE) and relative weight (Wr) of black crappie collected by frame netting, Lake Conroe, Texas, fall 1987-1993 (Sheet 1 of 3)



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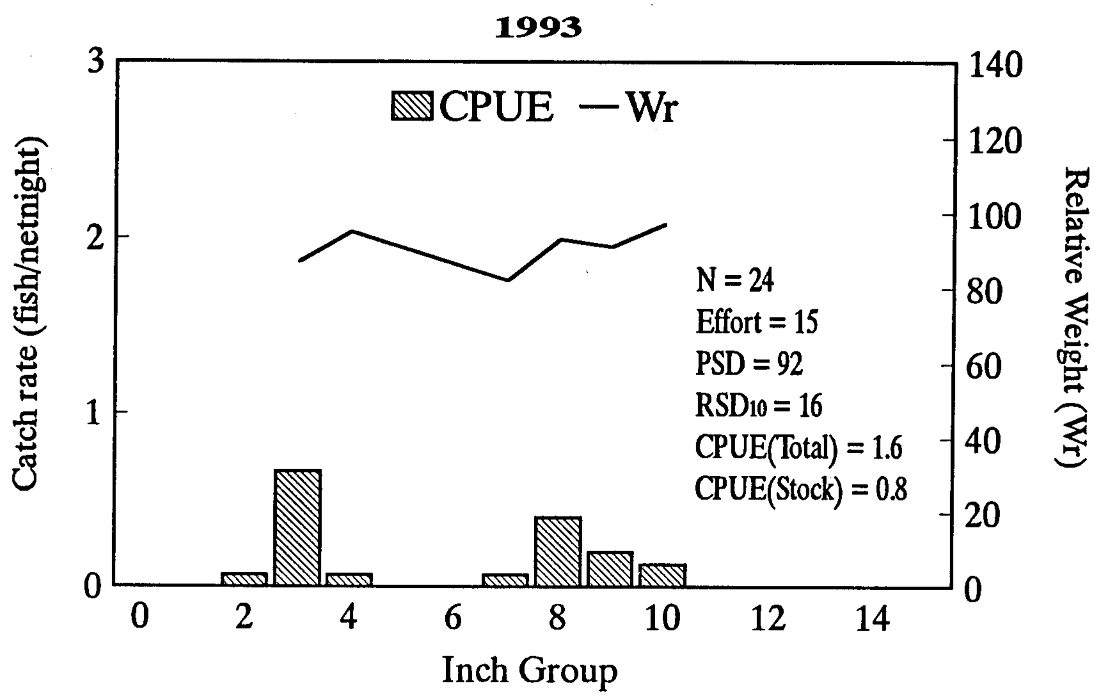

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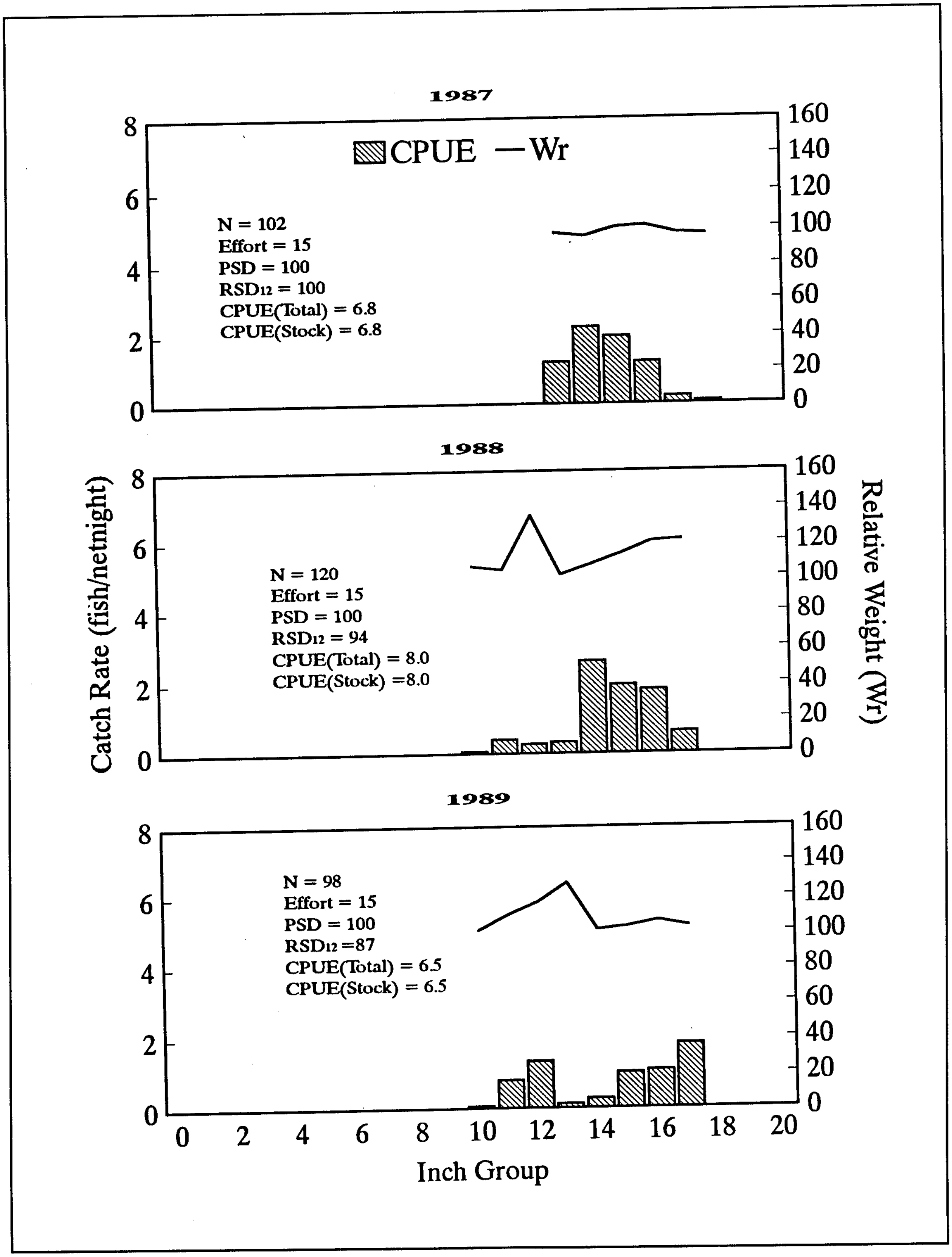
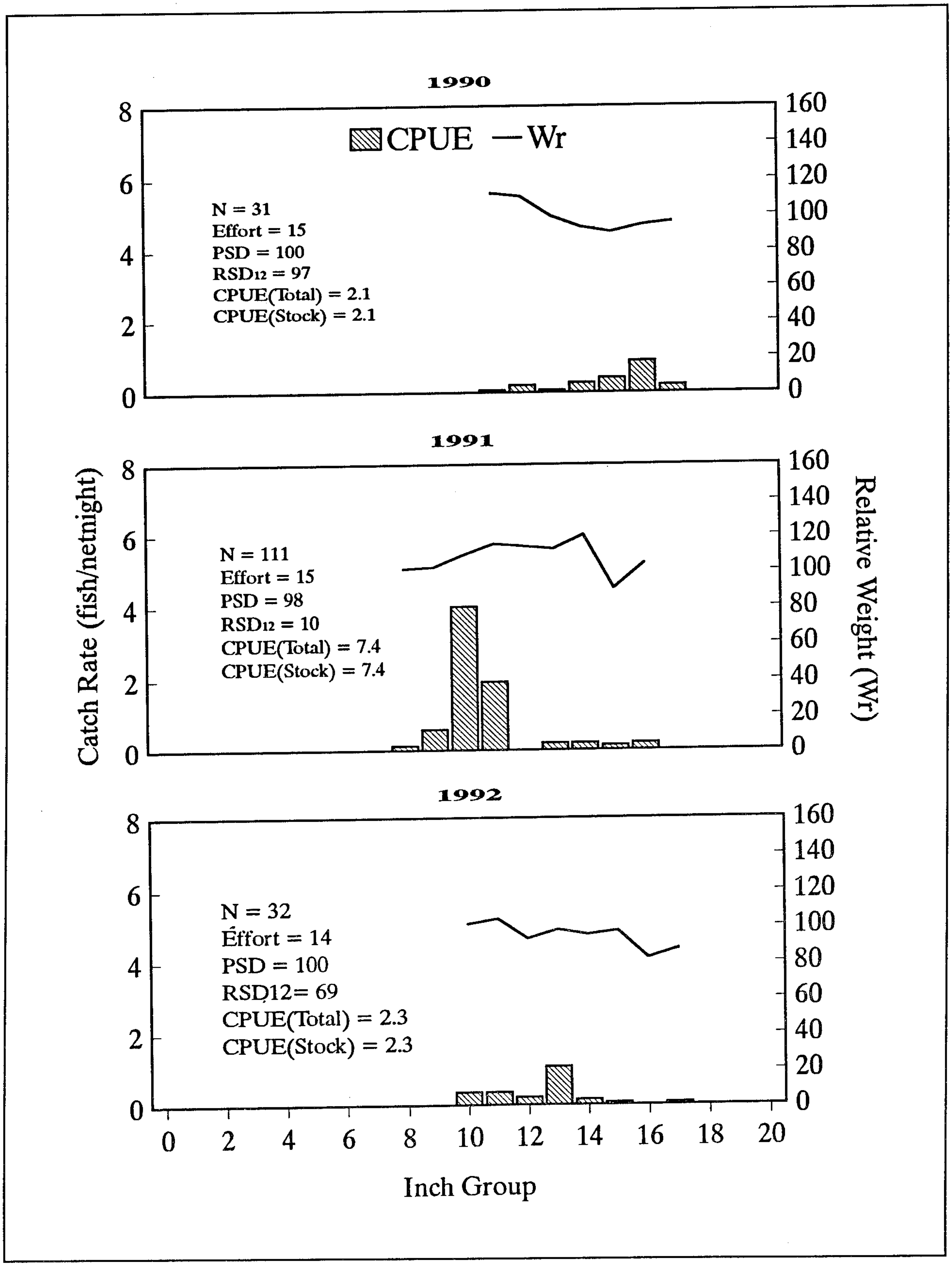


Figure 6. Catch per net night (CPUE) and relative weight (Wr) of white bass collected by gill netting, Lake Conroe, Texas, spring 1987-1993 (Sheet 1 of 3)



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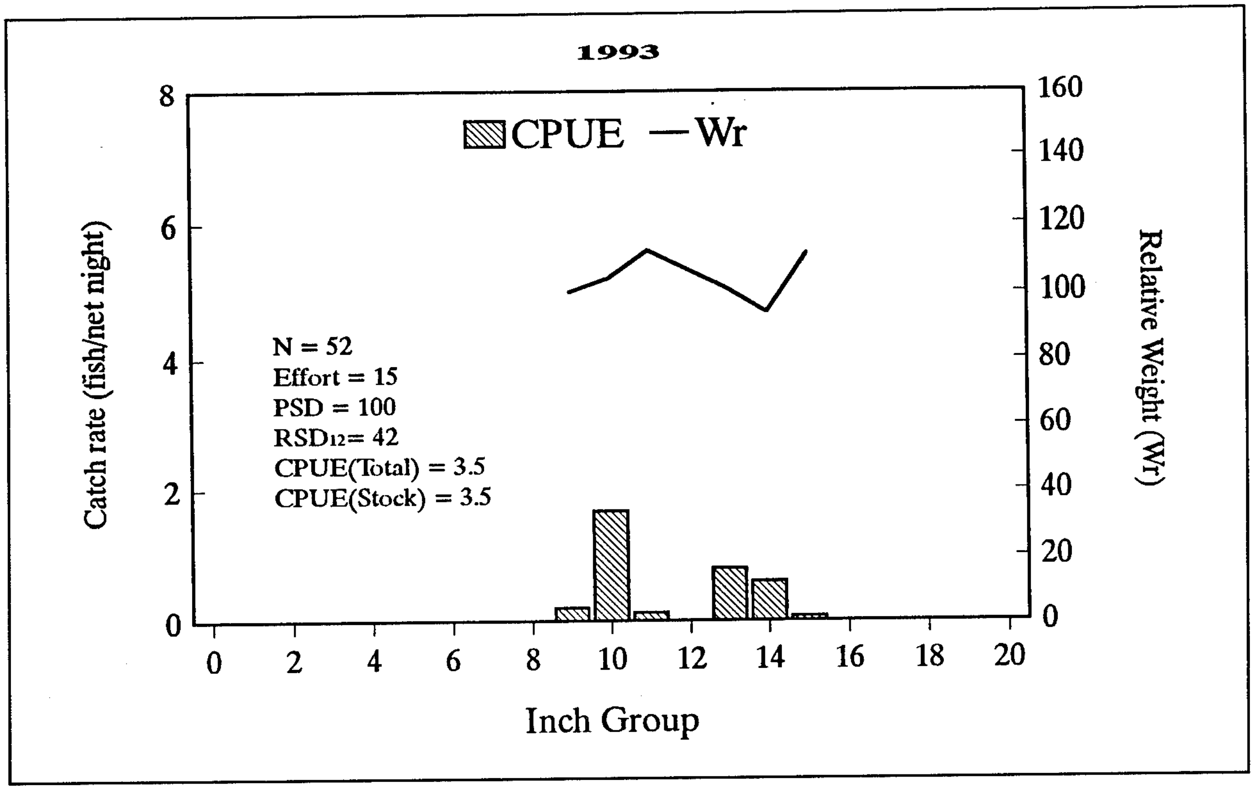

Figure 6. (Sheet 3 of 3)

Discussion

The effects of macrophyte removal on the Lake Conroe fisheries remain controversial. Whether fishing at Lake Conroe is better or worse than it was at the peak of macrophyte coverage is largely a matter of opinion. What is certain is that Lake Conroe is a high profile fishery near a major metropolitan area and that it is managed using every available tool. The Lake Conroe fisheries are still in a state of flux as productivity changes, species expand and decline, shoreline structure is altered by development, and management activities such as length limits and stockings take effect. The population information presented in this manuscript reflects all these factors and cannot be evaluated simply in terms of the effects of macrophyte loss.

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Stocking Update and Vegetation Changes in Lake Marion, South Carolina

by
Steven J. de Kozlowski[1]

Lake Marion, a 110,000-acre impoundment, was originally built and managed by the South Carolina Public Service Authority (a.k.a. Santee Cooper) as a multipurpose reservoir to generate hydroelectric power, improve commercial navigation, provide flood control, and provide recreational opportunities. Together, Lake Marion and Lake Moultrie comprise the Santee Cooper Lakes, one of the largest (170,000 acres) and most economically important lake systems in the southeast.

Historically, Lake Marion has not had a significant submersed aquatic plant population. However, that all began to change in 1982, when hydrilla was discovered growing near a fish camp in the upper portion of the lake. Despite a concerted effort to control this new infestation with aquatic herbicides, hydrilla spread and quickly replaced Brazilian elodea as the dominant submersed species. By 1987, it became obvious that hydrilla was rapidly spreading into areas that never before had submersed aquatic vegetation, was impacting a growing number of water uses, and threatened to infest up to one-half of the lake system.

In 1987, the South Carolina Aquatic Plant Management Council, a multiagency board that coordinates aquatic plant management activities in public waters, studied the problem and developed a management plan that recommended the use of triploid grass carp to provide long-term control of hydrilla. The plan called for the stocking of 300,000 sterile grass carp over a 3-year period with a final stocking rate of 25 fish per vegetated acre. A total of 100,000 grass carp would be stocked each year for 3 consecutive years in the upper lake area where hydrilla was most concentrated. While the stocking project was initially planned to run from 1989 to 1991, an additional 100,000 fish were released in 1992 to compensate for losses in 1989 from Hurricane Hugo and a spring fish kill event. Studies to monitor aquatic plant coverage, native fish populations, grass carp movement, and water quality changes were conducted by the U.S. Army Engineer Waterways Experiment Station.

Aquatic plant coverage was carefully documented prior to the stocking and was monitored annually following the stocking by infrared aerial photography and ground surveys. Following the first 2 years, hydrilla coverage continued to increase in the target area. However, by the third year, 1991, a moderate decline (20 percent) in submersed plant growth was apparent based on photointerpretation. Hydrilla coverage declined significantly (58 percent) in 1992, opening up areas of the lake that had been inaccessible to the public for many years. In 1993, hydrilla coverage was further reduced by about 10 percent. Upper Lake Marion, which used to be a popular fishing area because of its diverse aquatic habitat of open-water flats and flooded cypress/tupelo stands, is once again being heavily utilized by the public.

It is important to note that grass carp have not completely eliminated hydrilla or the other submersed aquatic species from the target area. Hydrilla, while substantially diminished, is still present throughout the upper lake. We have also seen an increase in the diversity of the submersed plant community and an increase in coverage of fanwort (*Cabomba caroliniana*), a native species.

1 Chairman, South Carolina Aquatic Plant Management Council, South Carolina Water Resources Commission, Columbia, SC.

Although grass carp have effectively controlled hydrilla in upper Lake Marion, hydrilla continues to spread throughout the system. A 1993 survey by Santee Cooper indicated that grass carp were responsible for controlling about 9,000 acres of hydrilla; however, hydrilla continued to impact an additional 17,000 acres in the lower portion of Lake Marion. In Lake Moultrie, where submersed plant coverage was less than 200 acres in 1989, hydrilla now covers almost 17,000 acres. As hydrilla continues to spread, water-use impacts are more frequent and more costly. In 1991, hydrilla was responsible for shutting down the St. Stephen Hydroelectric Project for several weeks by clogging intake screens. The estimated cost of lost power production was $90,000 per day. The shutdown also led to one of the largest fish kill incidents in the State's history with a replacement cost for lost game fish estimated at $526,600. Hydrilla now occurs in over 43,000 acres systemwide where it still impacts recreational uses and threatens two hydroelectric projects and a new water supply system.

Based largely on the success of the stocking project in upper Lake Marion, Santee Cooper and the Aquatic Plant Management Council have recently completed a long-term aquatic plant management plan for the lake system. The plan recommends use of an integrated control strategy using sterile grass carp for long-term control and aquatic herbicides for short-term control in high-use areas when control by grass carp is insufficient. Stocking rates will be 15 fish per vegetated acre with an upper limit of three fish per total surface acre (510,000 fish). The long-term plan was initiated in 1993 with the release of 50,000 grass carp into Lake Moultrie. The 1994 South Carolina Aquatic Plant Management Plan calls for the release of an additional 150,000 grass carp into Lake Moultrie and up to 5,000 grass carp into lower Lake Marion. To help underscore its commitment to this plan, Santee Cooper is completing a sterile grass carp hatchery located between the two lakes to supply fish for future years.

Lake Marion and Lake Moultrie are highly managed lakes that support a variety of important water uses. Sterile grass carp have proven effective in controlling hydrilla infestations in upper Lake Marion for the past 2 years without eliminating all aquatic plants in the system. The State has proceeded cautiously in this effort by stocking the fish incrementally, implementing conservative stocking rates, coordinating all activities with appropriate State and Federal agencies, and monitoring the results. We believe that an integrated management strategy that includes the use of sterile grass carp is the most environmentally and economically sound method of managing hydrilla in large lake systems like the Santee Cooper Lakes, and will ensure the continued viability of this important water resource.

Movements and Habitat Use of Triploid Grass Carp in Lake Marion, South Carolina

by

Jeffrey W. Foltz,[1] *J. Philip Kirk,*[2] *and K. Jack Killgore*[2]

Introduction

Lake Marion is a 44,000-ha impoundment composed of open water as well as dense cypress swamps. Aquatic vegetation had become a serious problem in upper Lake Marion north of the Interstate-95 bridge (I-95) (Inabinette 1985). Upper Lake Marion's shallowness is conducive to aquatic plant growth. Many areas of the lake had limited access because of dense aquatic vegetation. This hampered use of the lake by recreational hunters and anglers. Herbicides were used extensively in order to control the vegetation problem, but a more feasible and long-term solution was needed. Mechanical removal and herbicide treatment can be costly, time-consuming, and works on a limited basis for a limited time. One possible solution was the release of triploid grass carp (*Ctenopharyngodon idella*). Grass carp were effective in eliminating *Hydrilla verticillata* in several large lakes in Florida (Beach et al. 1976; Miley, Leslie, and Van Dyke 1979). Advantages of biological control, such as the use of triploid grass carp, include longevity of the method, constant fish-feeding activity against growing vegetation, low long-term cost, and high effectiveness on selected plants (Sutton and Vandiver 1986). A potential disadvantage of use of triploid grass carp would be overexploitation of aquatic vegetation and the consequent reduction in habitat for nongame fishes, other nongame wildlife, and waterfowl.

Three hundred thousand triploid grass carp were released between 1989 and 1991 at various locations in upper Lake Marion to control nuisance aquatic vegetation (South Carolina Aquatic Plant Management Council and South Carolina Water Resources Commission 1989). Subsequent supplemental stockings were being considered. Water temperature is known to cause migrational movement in grass carp once water reaches 15 to 17 °C (Aliev 1976). A rise in water level or increased flow rates have also caused grass carp to exhibit migrational movement (Stanley, Miley, and Sutton 1978). If triploid grass carp were to migrate up the rivers and away from aquatic weed-infested areas, they would be ineffective for weed control. Objectives of this study were to (a) determine the magnitude and direction of grass carp movements, (b) determine if grass carp remain in the targeted vegetation areas, and (c) examine characteristics of habitats utilized by triploid grass carp.

Study Site

Lake Marion, South Carolina, is a 44,000-ha lake formed by the impoundment of the Santee River. Lake Marion has an average depth of only 5 m and a maximum depth of 12 m. Immediately upstream of Lake Marion, the Santee River is formed by the confluence of the Wateree and Congaree rivers. The Wateree River originates at the Wateree Dam about 100 km upstream from Lake Marion. The Congaree River originates at the Saluda Dam on Lake Murray and flows 85 km before joining the Wateree. The Wateree and Congaree rivers average 183 m^3/sec and 266 m^3/sec discharge, respectively. Lake Marion is connected to Lake Moultrie by an unobstructed canal known as the diversion canal. Together, the two reservoirs comprise the "Santee-Cooper" system (Figure 1). When Lake Marion was constructed in 1941, it impounded

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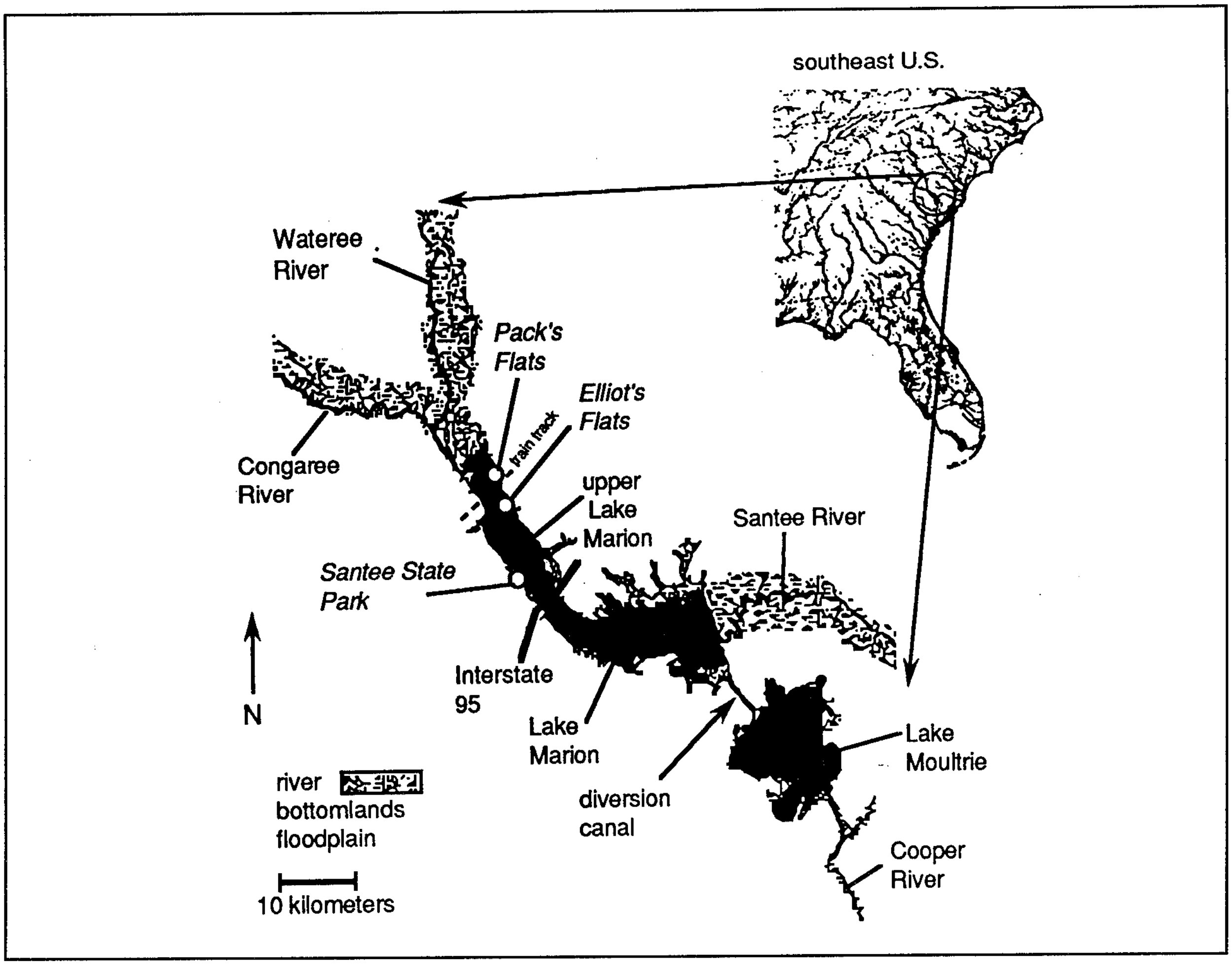

Figure 1. Study areas of upper Lake Marion in relation to the Santee-Cooper (Lakes Marion and Moultrie) system

6,500 ha in its headwater section, which was known as the Santee Swamp. The swamp is anaerobic most of the year and hence influences water quality in upper Lake Marion (Bates and Marcus 1989). Vegetated areas of the lake targeted for control have an estimated 4,800 ha of submerged vegetation, mostly upstream from the I-95 bridge. Habitats in upper Lake Marion can be categorized into five types: (a) Santee River channel (2 to 8 m deep), (b) open water with creek channels running through it (1 to 2 m deep), (c) open water with scattered cypress trees (2 to 3 m deep), (d) open-water shallow flats (1 to 2 m deep), and (e) thick cypress swamp (2 to 3 m deep). All types but the Santee River channel support dense stands of nuisance aquatic vegetation. No assessment of the proportion of upper Lake Marion that each habitat type comprises has been made. Likewise, no assessment of the magnitude of the swamp's influence on downstream water quality has been made, although fish kills because of anaerobic conditions are a common summer phenomenon in upper Lake Marion's swamps.

Methods

Eighty-two triploid grass carp were surgically implanted with radio transmitters (45 fish in the 1989/1990 study and 37 fish in the 1990/1991 study). Life spans of transmitters were 9 months, and transmitter frequencies ranged from 48.036 to 49.527 kHz. Each fish was identified by a distinct frequency. Fish were anesthetized using a bath containing 100-mg/L MS-222 and 25-mg/L Furacin. Each fish was weighed to the nearest 0.01 kg and measured

to the nearest millimeter. Anesthetized fish were placed in a V-shaped operating trough so that the fish excluding its abdomen was submersed in water containing MS-222 and Furacin. A small aquarium aerator was used to maintain adequate oxygen levels in the operating trough. A radio transmitter was then surgically implanted using the procedure described by Schramm and Black (1984). Surgical gloves were worn, and instruments and transmitters were disinfected prior to use.

Scales were removed from the incision area, and a 5-cm longitudinal incision was made in the ventral wall 6 cm anterior to the pelvic girdle. A transmitter was then inserted into the body cavity and the incision closed with non-absorbable silk sutures. Oxytetracycline (50-mg/kg body weight) was injected into the body cavity before the last suture was stitched. Fish were then immediately released into the lake at Pack's or Elliot's Flats (1989/90 study) or approximately 1 km north of Santee State Park during the 1990/91 study (Figure 1).

An Advanced Telemetry Systems (Model 2000) radio receiver was used. Boat searches for implanted grass carp were conducted 3 days per week for 18 months. Signals were received while boating with a Telex Communications (Model 64 B-S) four element yagi antenna. Once a signal was picked up by the receiver, the antenna was rotated to ascertain direction, and the boat was motored in that direction. Signal strength increased as the fish was approached. The coax cable was then disconnected from the antenna and dropped in the water beside the boat. Intensity of the signal indicated when the boat was within 25 m of the fish. Air searches of the Santee-Cooper system were conducted monthly during the initial study in order to locate any fish that had not been located in 14 days. Air searches were not conducted during the second study because all fish were regularly located.

Once a fish was located, date, water depth, and Loran latitude and longitude coordinates were taken. Water temperature and dissolved oxygen were measured at the surface and on the bottom with a YSI dissolved oxygen meter (Model 51B) to the nearest 0.1 °C and 0.1-mg/L, respectively. Mean dissolved oxygen was calculated from the surface and bottom value. An aquatic vegetation sample was taken from the surface and bottom with a rake. Aquatic vegetation was identified (Pennwalt 1984) in the field. The species that comprised the largest proportion of a sample was categorized as primary vegetation, and the species that comprised the next largest proportion was categorized as secondary. Vegetation density in the general vicinity of the fish location was categorized into one of four categories: (a) vegetation covers ≥50 percent of the surface, (b) vegetation covers <50 percent of the surface, (c) vegetation present but submersed, or (d) vegetation sparse. Habitat type was categorized as one of five types (see Study Site): (a) river channel (Santee, Congaree, or Wateree), (b) open water with creek channels (OWCC), (c) open-water shallow flats (OWSF), (d) thick cypress swamp (TCS), and (e) open water with scattered cypress trees (OWCS).

Fish locations were plotted on a digitized map that indicated the Santee River channel. Days elapsed and distance moved (to nearest 0.01 km) between readings were computed for each fish. Minimum net daily movement was then computed as net kilometers/elapsed days. Linear and nonlinear regressions were employed to describe seasonal changes in minimum net daily movement. Distance from the river channel to the fish was computed along a line perpendicular to the fish and the river channel. Distances were recorded to the nearest 0.01 km. Distance of grass carp from the river channel was tested with a t-test (null hypothesis that the distance = zero).

Core-use area and home-range estimates were calculated for individual fish using the bivariate scatterpoints (Longitude = Y_1 and Latitude = Y_2) of grass carp locations. Core-use areas were computed as the 95-percent confidence region (i.e., ellipse) to each fish's mean location, whereas home ranges were based on a 95-percent confidence region to each fish's observations (Sokal and Rohlf 1969). Average core-use and home-range

areas were computed from log10 transformed values. Similarly, the statistical center of distribution for both studies was computed as the 95-percent confidence region (i.e., ellipse) to the sample mean.

Results

Triploid grass carp used in this study averaged 704 mm total length (SE = 15) and 4.42-kg live weight (SE = 0.20) at the time of release. Fish were located on 225 occasions in the first study and 180 occasions during the later study. Average elapsed time between locations of an individual fish was 10 and 17 days for the first and second study, respectively. The longest distance moved by a fish was 10.6 km over 4 days, while the averages were 0.29 km/day (SE = 0.01) and 0.10 km/day (SE = 0.01) for the 1989/90 and 1990/91 studies, respectively. Grass carp in this study did not demonstrate a preference for the river channel in Lake Marion (Figure 1). Mean distance of grass carp from the river channel was 1.75 km (SE = 0.08) and 1.01 km (SE = 0.07) for the first and second studies, respectively. These mean distances was significantly different from zero ($t = 22.84$, $p = 0.0001$; $t = 14.21$, $p = 0.0001$). Grass carp in the 1989/90 study showed an average individual core-use area of 49 km^2 and an individual home range of 165 km^2. As a group, they were distributed from Pack's Flats to Santee State Park (Figure 2). Grass carp in the second study had an average individual core-use area of 9 km^2 and only 49 km^2 for average individual home range (Figure 2). As a group, fish in the second study were located principally in an area known as Stumphole Flats about 3 km north of Santee State Park (Figure 2). Surface dissolved oxygen concentrations at fish locations remained above 8 mg/L most of the year, but bottom concentrations averaged less than 1 mg/L in May. Water temperatures at fish locations were similar to the pattern that occurs in the upper lake: winter temperatures of 10 °C and summer high temperatures of 29 °C. There was a 2 to 5 °C difference between surface and bottom temperatures at fish locations, which ranged in depth from 2 to 3 m.

No studies to date have quantified the different proportions of habitat and aquatic vegetation types in upper Lake Marion. Fish were located in open-water shallow flats or adjacent open water-cypress stands 66 and 70 percent of the time during the 1989/90 and 1990/91 studies (Figure 3). Depths averaged 2 to 3 m. Grass carp locations in thick cypress swamps accounted for only 19 and 25 percent of the locations. Seventy-two and fifty-three percent of recorded grass carp locations were in areas with aquatic vegetation at the water's surface (Figure 4). Over the two studies, 66 and 70 percent of fish locations were in areas dominated by *Hydrilla* (Figure 5). *Egeria densa* was the predominate vegetation type in only 11 and 6 percent of fish locations. Other vegetation types that included duckweed, *Nitella*, and coontail accounted for 20 and 19 percent of the locations, but no single species exceeded 2 percent.

Discussion

Magnitude of movements were comparable but less than those reported for adult fish by Bain et al. (1990). Bain et al. (1990) reported that adult fish movement averaged 33 km over a 4-month period (i.e., about 0.27 km/day) and that one fish traveled 6 km/day. It is difficult to make direct comparisons between studies without knowledge of the frequency of observation. For example, a fish could travel 1 km each day for 10 successive days, but if it finished at its origin and had not been observed for 10 days, net daily movement would compute as zero. Higher movement rates reported for the first study (1989/90) were for grass carp released at two widely separated points in upper Lake Marion, South Carolina. Also, fish probably moved to avoid widespread low dissolved oxygen events that occurred during 1989 and 1990 in the uppermost cypress swamps and adjacent flats. In the 1990/91 study, grass carp generally remained in the shallow flats located within 2 km of their release site. Core-use areas traveled by grass carp were very close to values reported by Chilton (in this symposium) for triploid grass carp in a Texas reservoir.

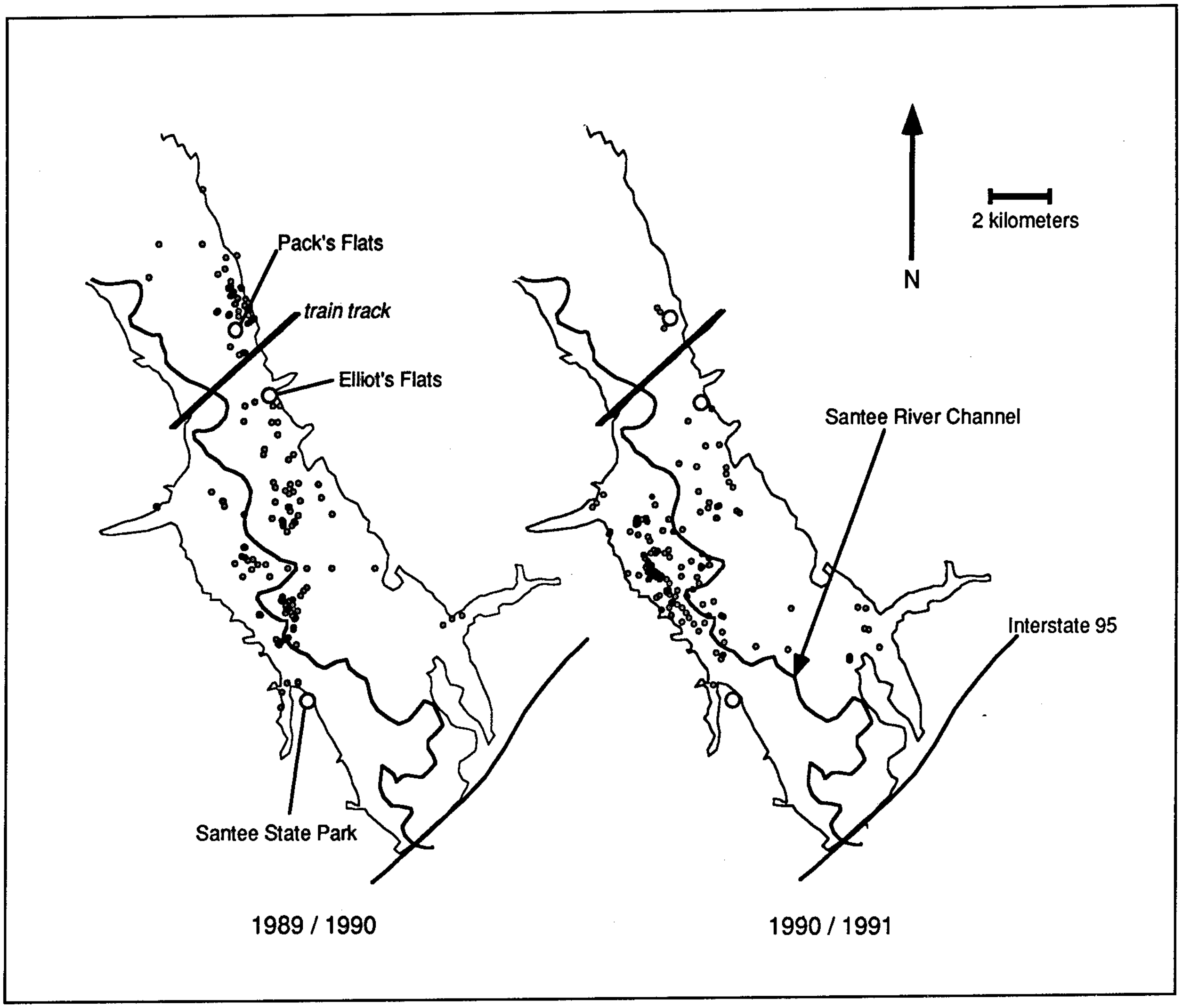

Figure 2. Locations of radio-tagged adult triploid grass carp in upper Lake Marion, South Carolina. Each dot represents one or more individual fish locations

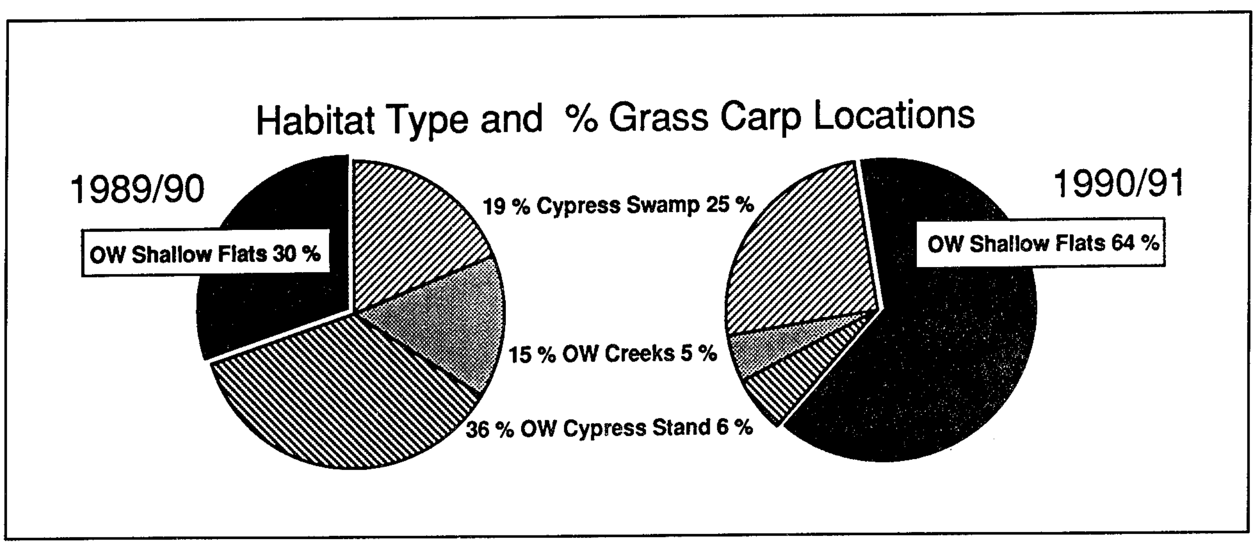

Figure 3. Habitats utilized by radio-tagged adult triploid grass carp in upper Lake Marion, South Carolina

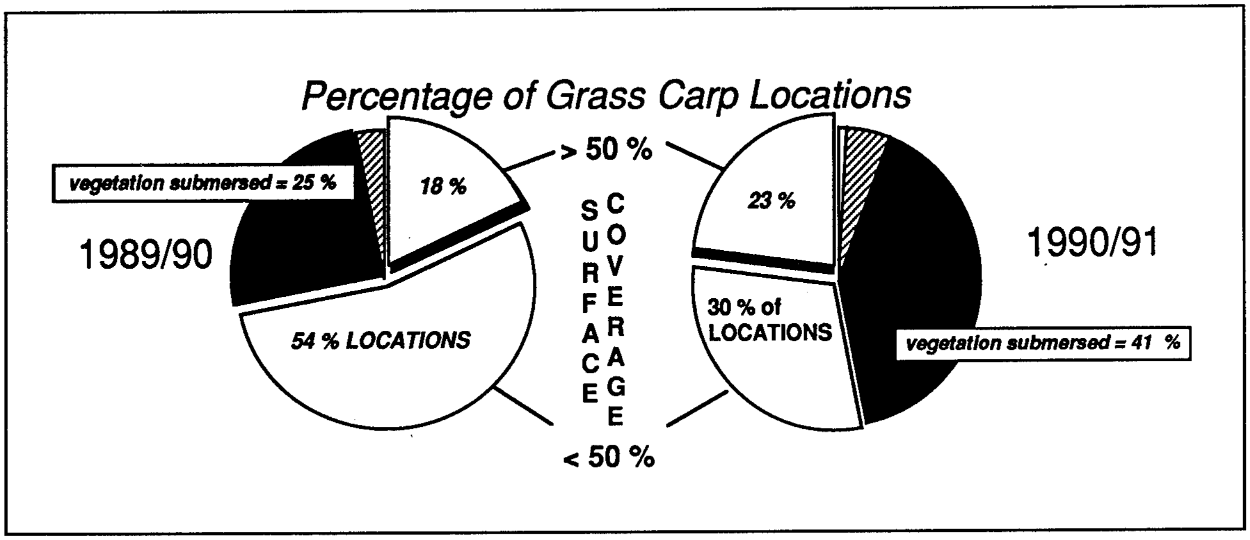

Figure 4. Aquatic vegetation relative abundances at locations utilized by radio-tagged adult triploid grass carp in upper Lake Marion, South Carolina

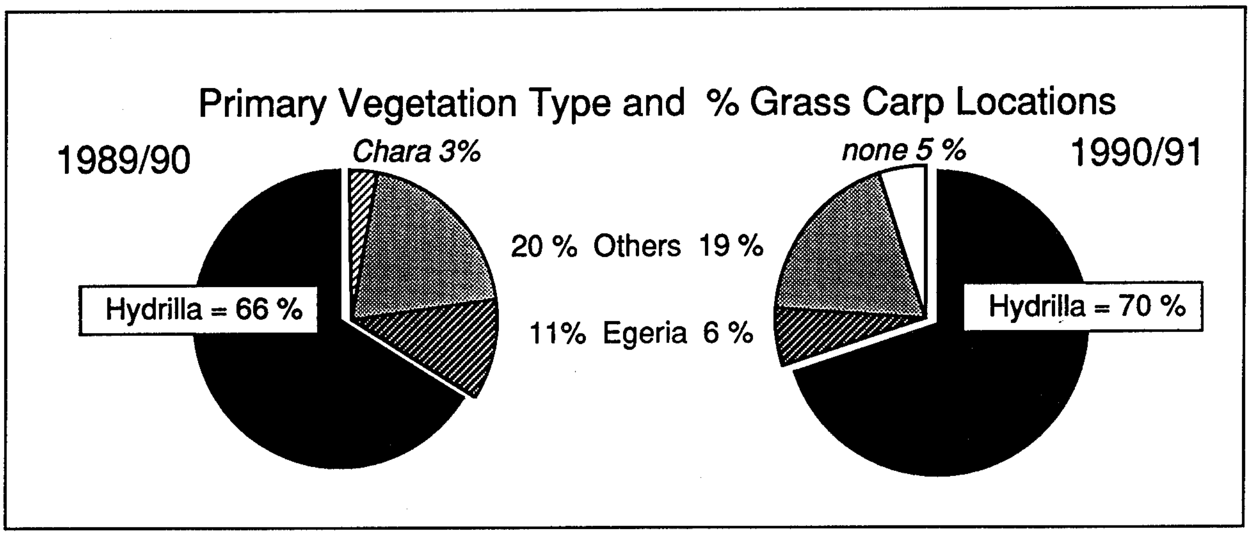

Figure 5. Predominate aquatic vegetation at locations utilized by radio-tagged adult triploid grass carp in upper Lake Marion, South Carolina

Shallow flats and adjacent areas were most frequented by grass carp, probably because of higher dissolved oxygen and abundant submersed vegetation. Thick cypress swamps in contrast are characterized by low summer dissolved oxygen concentrations (Bates and Marcus 1989). Utilization of more open areas probably provides grass carp with a suitable combination of food density and dissolved oxygen concentrations.

Locations predominated by *Hydrilla* constituted the majority of grass carp locations. *Hydrilla* is an excellent food for grass carp because of the soft nature of the plant and high ash content (Tan 1970; Rottman 1977). Grass carp used in this study were large (704-mm TL), and according to Sutton and Vandiver (1986), *Hydrilla* would be their preferred food. No data exist concerning the percentage of Lake Marion's total nuisance aquatic weeds that the individual species comprise. Thus, no preferences for *Hydrilla* can be inferred in the present study.

Summary

No long distance migrations were observed, and fish showed no affinity for the Santee River channel. Triploid grass carp remained in the upper part of Lake Marion, and these fish did not leave areas targeted for aquatic vegetation control. Dissolved oxygen levels appeared to play an important part in the location and movement of fish. In addition, water quality and aquatic vegetation distribution for upper Lake Marion are needed in order to interpret movements relative to available habitat.

Acknowledgments

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Lake Guntersville Case Histories

Aquatic Vegetation in Guntersville Reservoir Following Grass Carp Stocking

by

David H. Webb,[1] Larry N. Mangum,[1] A. Leon Bates,[1] and H. Douglas Murphy[2]

Introduction

Guntersville Reservoir is a large (27,500-ha, 122-km-long) impoundment of the Tennessee River in northeastern Alabama and southeastern Tennessee and is one of several impoundments operated by the Tennessee Valley Authority (TVA). The reservoir is a multipurpose project that provides for navigation, flood control, power production, recreation, and other uses. Dams at the downstream and upstream ends of the reservoir have hydropower generation units and locks for commercial navigation.

Several exotic submersed macrophytes such as Eurasian watermilfoil (*Myriophyllum spicatum*), spinyleaf naiad (*Najas minor*), hydrilla (*Hydrilla verticillata*) and native species such as southern naiad (*Najas guadalupensis*), coontail (*Ceratophyllum demersum*), American pondweed (*Potamogeton nodosus*), small pondweed (*Potamogeton pusillus*), muskgrass (*Chara zeylandica*), and other aquatic macrophytes have created reservoir-use conflicts requiring control in several areas of Guntersville Reservoir. The TVA uses drawdowns and herbicides to control nuisance populations of aquatic macrophytes along developed shorelines, marinas, public-use sites, and commercial recreation areas on Guntersville Reservoir and several other reservoirs in the Tennessee River system (TVA 1994).

In the late 1980s, aquatic macrophytes colonized about 8,200 ha or about 29 percent of the reservoir's surface area. Hydrilla, which was first discovered in Guntersville Reservoir in 1982, colonized about 1,160 ha in 1988 and was considered a long-term threat to multiple uses of Guntersville Reservoir, as well as several other reservoirs within the TVA system.

As a result of public concerns, TVA and the U.S. Army Corps of Engineers were asked to develop a 5-year plan for reducing submersed aquatic plants in Guntersville Reservoir and to develop and demonstrate more efficient and effective methods of managing aquatic vegetation (Bates, Decell, and Swor 1991). The project, referred to as the Joint Agency Guntersville Project (JAGP), was initiated in 1990 and will continue through 1994.

Grass Carp Demonstration

One component of the JAGP was a large-scale demonstration with grass carp to control hydrilla and reduce populations of other aquatic macrophytes. After preparation of an environmental assessment (TVA 1990), TVA released 100,000 triploid grass carp in Guntersville Reservoir from April to July 1990. This release was equivalent to about 17 fish/vegetated hectare based on aquatic macrophyte coverage measured in 1989. An additional 20,000 grass carp were released in Guntersville Reservoir from 1987 to 1989, primarily by private citizens or lake user groups. Thus, density of grass carp following the stocking of the 100,000 fish in 1990 was estimated to be about 20 fish per vegetated hectare. Incremental stocking of grass carp was planned for the demonstration; but because of unexpected declines in vegetation coverage during the first year of the study, no additional grass carp were released by TVA.

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1 Tennessee Valley Authority, Clean Water Initiative, OSA 1B, Muscle Shoals, AL.
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A monitoring program began in 1990 to document changes in coverage and composition of aquatic macrophyte communities to provide an assessment of grass carp herbivory and to provide supportive data for other assessments, such as fisheries (see Wrenn et al., this volume) and other projects associated with the JAGP. A summary of aquatic macrophyte monitoring results for 1990 to 1992 can be found in Webb (1993).

Prior to this demonstration, studies with grass carp were conducted on Guntersville Reservoir by TVA in a 160-ha screened embayment (TVA 1987, 1989). Bain (1993) also studied grass carp movement within Guntersville Reservoir. While these studies provided a basis for the grass carp demonstration, the numerous uncertainties associated with use of grass carp in large systems (Nobel, Bettoli, and Betsill 1986; Bain 1993) required a reservoir-scale approach to address various questions. Knowledge from the JAGP demonstration will be used to evaluate whether grass carp should be used in other reservoirs within the TVA system.

Materials and Methods

Quantitative sampling of aquatic macrophyte communities has been conducted since 1990 at several locations in Guntersville Reservoir from Tennessee River Mile (TRM) 356.0 to TRM 394.2. Aerial photography has been acquired annually since the late 1970s by TVA's aquatic plant management program and is used as supportive baseline data. Representative data from selected sampling sites (Figure 1) have been included in this report with a generic description of methods.

Coverage of aquatic macrophytes was determined from large-scale (1:7200), color aerial photography acquired annually during September or early October when biomass is typically at its peak. Aquatic macrophyte colonies were delineated on mylar overlays attached to the photographic prints and labeled by species. Area of delineated colonies was obtained using an electronic planimeter.

Aboveground biomass was collected using an open-ended plexiglass box sampler with a sampling area of 0.25 m^2, a hydraulically operated aquatic plant sampler developed for the U.S. Army Corps of Engineers Aquatic Plant Control Research Program (Sabol 1984) having a sampling area of 0.39 m^2, or with a mechanical harvester with a cutting head width of 1.6 m. The box sampler was used in shallow-water sites that were generally less than 1 m deep. The hydraulically operated sampler and the mechanical harvester were used to sample deeper water sites (1 to 4 m). Wet weights of plant samples were obtained after spinning the samples in a washing machine for 6 min.

The aquatic plant community at many of the shallow-water sites frequently was composed of a mixture of species such as spinyleaf naiad, southern naiad, muskgrass, small pondweed, and horned pondweed (*Zannichellia palustris*). Because of the excessive amount of the time required to separate individual species, wet weights for this group of plants were determined collectively for each sample. They have been labeled annuals for purposes of this report, although species such as horned pondweed may be a perennial plant. Species such as Eurasian watermilfoil, hydrilla, American pondweed, and coontail were individually separated and weighed, but their wet weights occasionally were summed.

Exclosures were constructed in the late winter or early spring of 1992 and 1993 from wire or block-nets having openings or mesh size small enough to exclude grass carp. Size of exclosures varied from 1 m^2 to as large as 2 ha.

Results

Coverage of submersed aquatic macrophytes on Guntersville Reservoir almost doubled from about 4,360 ha in 1984 to 7,830 ha in 1988 during a period of record drought in the Tennessee Valley (Figure 2). Lowest macrophyte coverage was about 2,010 ha in 1991, the second year after the large-scale grass carp

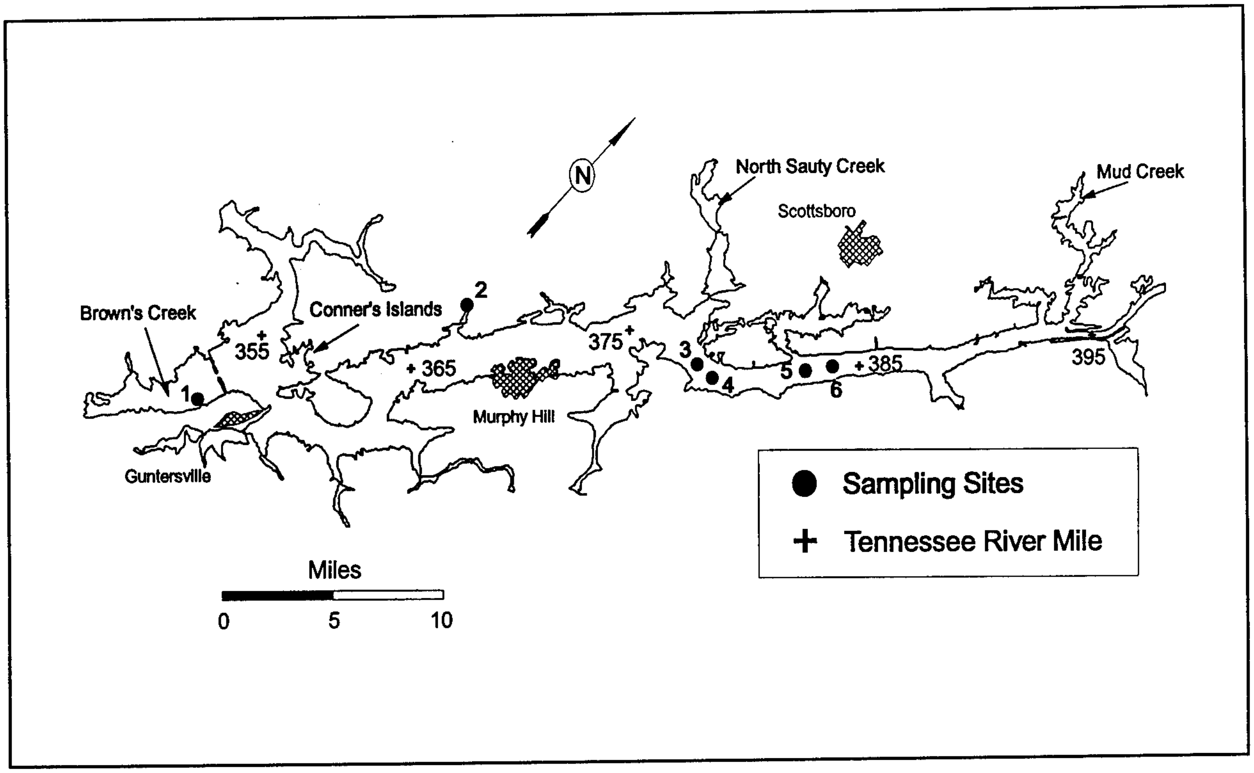

Figure 1. Locations of selected aquatic macrophyte sampling sites on Guntersville Reservoir (1 = Brown's Creek, 2 = Mill Creek, 3 = Powerline Milfoil, 4 = Powerline Hydrilla, 5 = Chisenhall Shallow, 6 = Brewster Hydrilla)

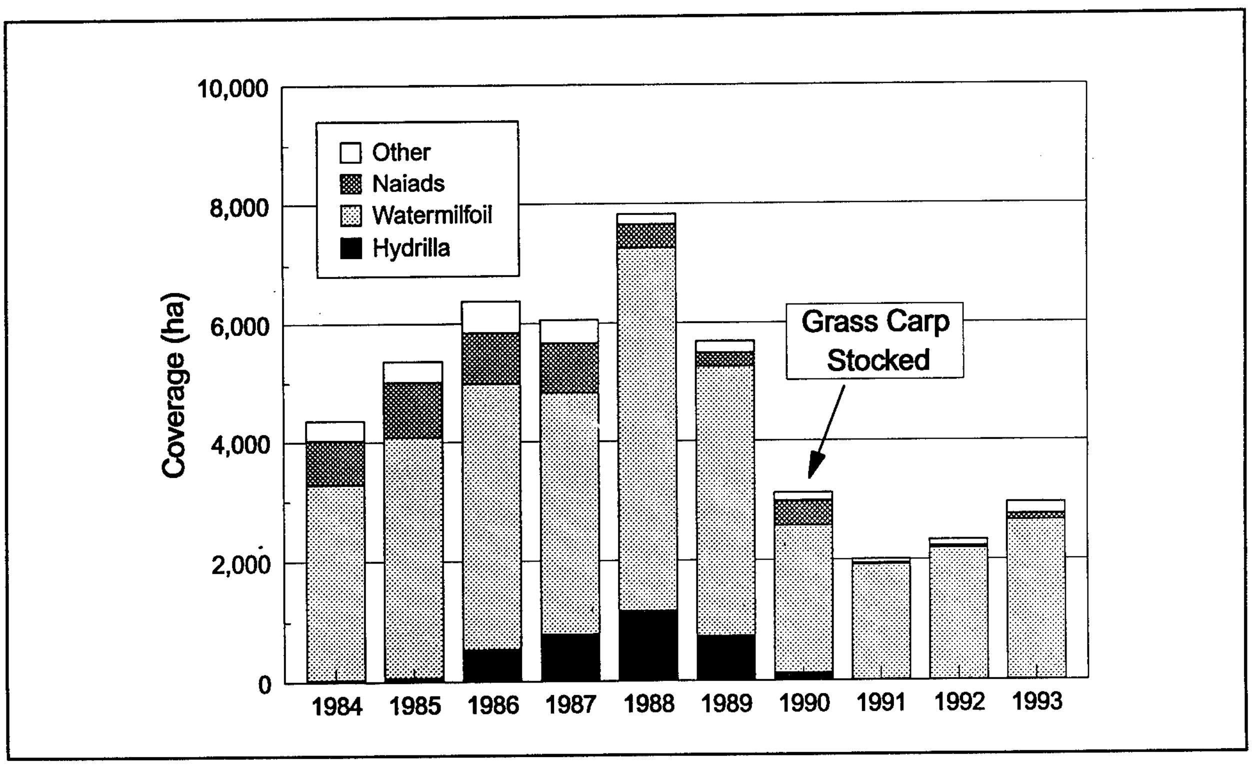

Figure 2. Submersed aquatic macrophyte coverage determined from aerial photography on Guntersville Reservoir from 1984 to 1993

stocking. Since 1991, macrophyte coverage has increased on Guntersville Reservoir and was about 2,960 ha in 1993.

Since the stocking of grass carp, coverage of several species such as spinyleaf naiad, southern naiad, small pondweed, muskgrass, and hydrilla declined, then showed evidence of regrowth. Naiads, which colonized about 410 ha in 1990, declined to less than 30 ha in 1991, but increased to about 100 ha in 1993. Since 1991, colonies of naiads, muskgrass, and narrow-leaved pondweeds generally occurred as small colonies in very shallow water or in shallow subembayments that were isolated from the main reservoir.

Hydrilla, which colonized about 750 ha in 1989 (the year prior to the large-scale grass carp stocking), declined to about 130 ha in 1990 and to less than 1 ha during subsequent years (Figure 2). The 130 ha of hydrilla in Guntersville Reservoir in 1990 was primarily in overbank habitats along the old river channel from TRM 378 to TRM 384. The only hydrilla colonies visible at the surface in Guntersville Reservoir in 1991 and 1992 were about 1 ha in coverage and occurred as a narrow band in the vicinity of TRM 390.5. This colony and another in a backwater slough in the same vicinity were "topped out" in 1993. Widely scattered plants of hydrilla also were visible in 1993 at the surface at TRM 368.7 at the downstream end of a submersed island (Pine Island) that was colonized by more than 100 ha of hydrilla in 1988.

Eurasian watermilfoil coverage was about 80 percent of the total submersed macrophyte coverage in 1989 and 1990 and about 90 to 95 percent of the total in 1991, 1992, and 1993 (Figure 2). From 1990 to 1991, several large colonies of Eurasian watermilfoil "disappeared" from the portion of the reservoir downstream of TRM 373. Beginning in 1992, Eurasian watermilfoil recolonized some overbank habitat from TRM 363 to 373, but is still sparse in downstream overbank habit and embayments from TRM 354 to 358 where it was abundant in 1990.

A decline in submersed aquatic macrophyte coverage comparable with Guntersville Reservoir occurred in other mainstream TVA reservoirs such as Kentucky, Wheeler, and Chickamauga reservoirs from 1988 to 1990 (Figure 3). The increase in macrophyte coverage from 1984 to 1988 was related to optimum growth conditions (low flow and clear water) associated with record drought years, while the decline from 1988 to 1990 was caused largely by high flows and associated factors. Since 1990, submersed aquatic macrophytes have increased on these reservoirs and, with the exception of Chickamauga, exhibit trends comparable with Guntersville Reservoir. The increase corresponds to the return to more normal flow and growth conditions in the Tennessee Valley.

Biomass and Community Composition

A comparison of biomass from 1990 to 1992 of various macrophyte species from shallow-water sampling sites showed an almost total loss or significant decline in spinyleaf naiad, southern naiad, muskgrass, and small pondweed, which typically dominate the annual aquatic plant community. This occurred not only in Mill Creek embayment (Figure 4), but at other shallow-water sampling sites in North Sauty Creek and Mud Creek embayments. In 1993, the annual plant community increased at the Mill Creek site (Figure 4) and at the sample site in Mud Creek embayment. In North Sauty Creek embayment that had a mean depth of about 1 m, regrowth of annuals did not occur, and Eurasian watermilfoil occurred in monospecific stands.

Field observations from other portions of Guntersville Reservoir during 1990 to 1992 indicated a similar reservoirwide trend for annual plants in shallow-water habitat. In 1991 and 1992, the annual plant community was almost totally absent except in shallow-water habitats that were only a few centimeters deep and in small coves that had beaver dams or other physical barriers to grass carp movement.

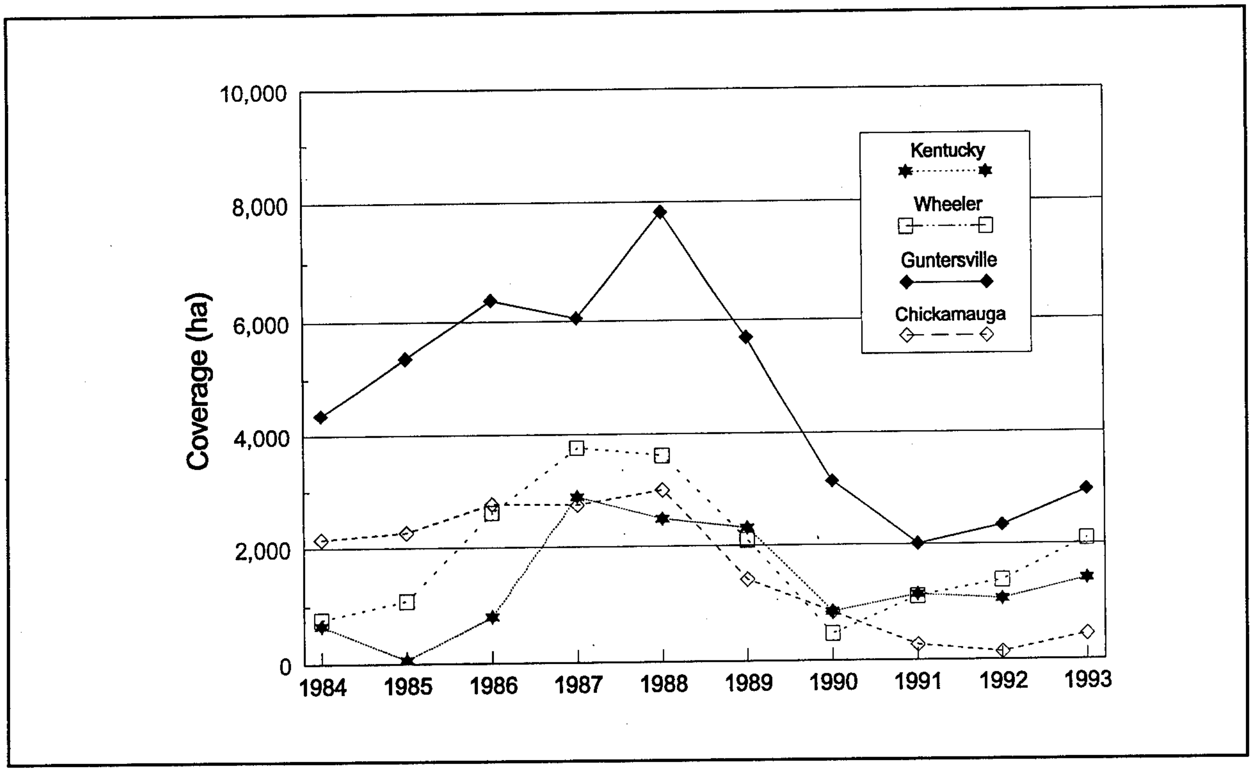

Figure 3. Aquatic macrophyte coverage determined from aerial photography on Kentucky, Wheeler, Guntersville, and Chickamauga Reservoirs from 1984 to 1993

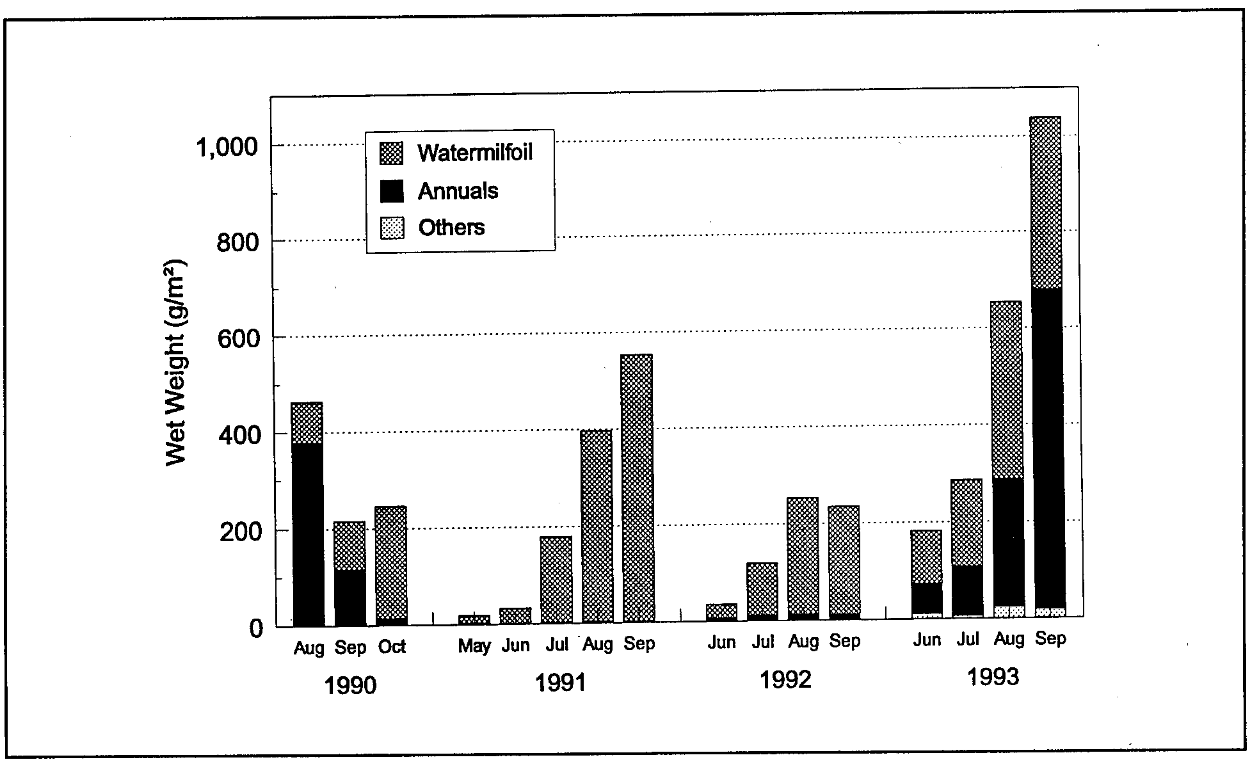

Figure 4. Mean aboveground biomass of aquatic macrophytes at the Mill Creek sampling site from 1990 to 1993

Regrowth of annual species, as was the case in Mill Creek embayment, was observed in shallow-water habitat in 1993 at other scattered localities within the reservoir in slightly deeper water (i.e., 0.5 m).

Peak biomass at sampling sites in deeper water (1 to 2.5 m) from TRM 378.5 to TRM 384 that were colonized by almost monospecific colonies of Eurasian watermilfoil in 1990 (Powerline Milfoil site) remained relatively stable (Figure 5). In contrast, sampling sites dominated by hydrilla in 1990 (Brewster Hydrilla site) had significant declines in this species (Figure 6). Eurasian watermilfoil became the dominant submersed macrophyte at these sites and was present in near monospecific stands by the end of the 1991 growing season. At two of the sampling sites dominated by hydrilla in 1990, Eurasian watermilfoil became the dominant macrophyte in 1991, and peak biomass declined from 3.566 kg/m^2 at the Powerline Hydrilla site in 1990 to 1.263 and 0.904 kg/m^2 in 1991 and 1992, respectively, and from 2.725 kg/m^2 at the Brewster Hydrilla site in 1990 to 1.572, 1.771, and 1.814 kg/m^2 in 1991, 1992, and 1993, respectively. Although hydrilla was less than 0.3 percent of peak total biomass at the Brewster Hydrilla site from 1991 to 1993, it still occurred in 64 to 100 percent of monthly samples in 1991, 0 to 4 percent in 1992, and from 16 to 40 percent of monthly samples in 1993. Most hydrilla samples consisted only of stems and rhizomes a few centimeters in length.

While hydrilla almost totally declined in overbank areas from TRM 378 to TRM 384 from 1990 to 1991, hydrilla remained in 1991 in 1,000-m^2 exclosures that were constructed to exclude grass carp in order to conduct insect biocontrol studies with *Hydrellia pakistanae* (Grodowitz and Snoddy 1992).

Exclosures

With few exceptions, there were significant differences in biomass and/or species composition of aquatic macrophytes within exclosures. Biomass within the 27.4- by 27.4-m exclosure at Brown's Creek in 1993 ranged from 250 to 398 g/m^2 for monthly sampling June through August compared with no plants outside the exclosure (Figure 7). Eurasian watermilfoil was 100 percent of the biomass within the exclosure during all months. The Brown's Creek sampling site was located in the downstream portion of Guntersville Reservoir where only minimal amounts of submersed aquatic macrophytes have occurred since 1991.

In a 30.5- by 76.2-m block-net exclosure site (Chisenhall Shallow) in an upstream area of the reservoir where Eurasian watermilfoil is abundant and widespread, total biomass (2,111 g/m^2) inside the exclosure during the peak month (August) was not significantly different from total biomass (1,525 g/m^2) outside the exclosure (Figure 8). However, species composition was significantly different with only 7 g/m^2 of annual biomass outside the exclosure compared with 1,580 g/m^2 inside the exclosure. Similar patterns of significantly greater biomass and/or differing species composition occurred inside versus outside exclosures in Brown's Creek and Mud Creek embayments and in Murphy Hill Cove where wire fences excluded grass carp from areas ranging in size from 0.3 to 2 ha.

Discussion

Submersed macrophyte coverage on Guntersville Reservoir declined from about 5,690 ha in 1989 (the year prior to grass carp stocking) to about 3,130 ha in 1990 (the year of stocking) to a low of 2,010 ha in 1991. Since 1991, macrophyte coverage has increased to 2,960 ha. Similar declines and a subsequent increase in macrophyte coverage occurred on most other mainstream reservoirs within the TVA system. Thus, much of the change in macrophyte coverage on Guntersville Reservoir is likely related to climatic conditions and associated factors (e.g., water clarity, flow, and temperature) that can greatly influence macrophyte populations (Smith and Barko 1990).

Biomass in shallow-water habitat of embayments and deeper (1 to 2.5 m) overbank

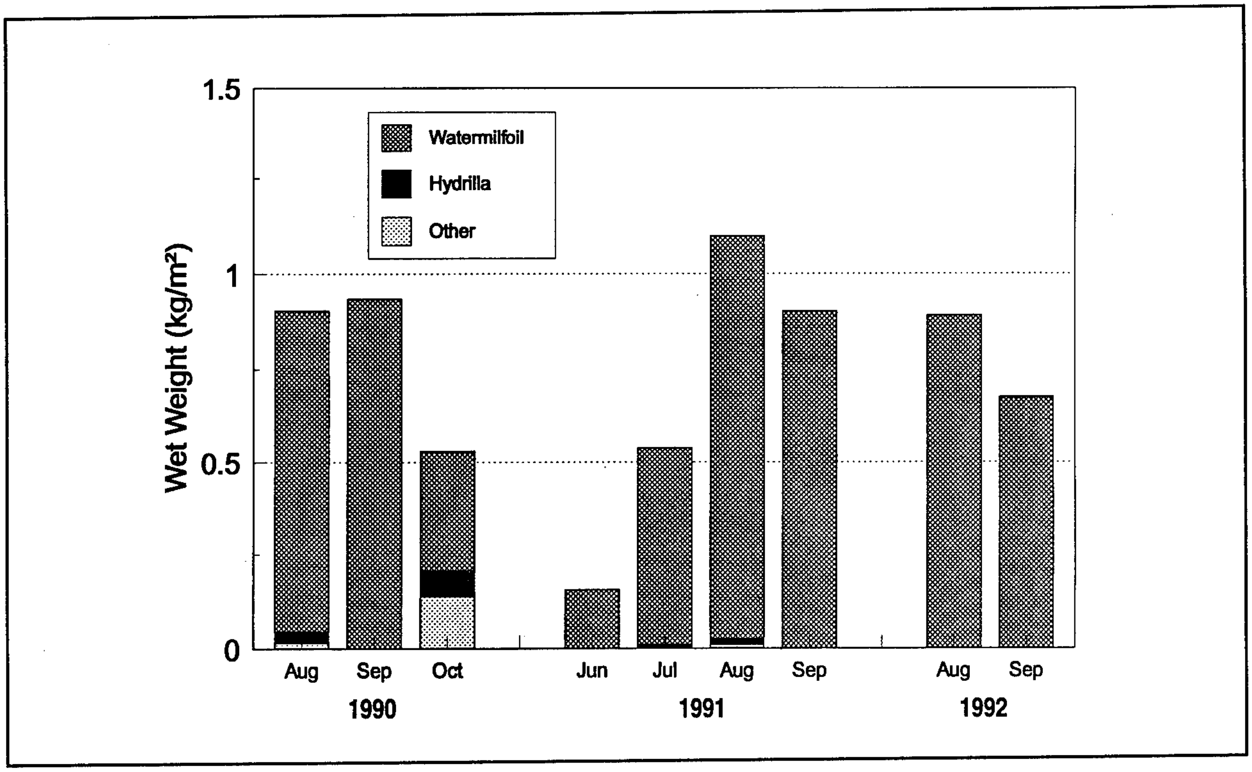

Figure 5. Mean aboveground biomass of aquatic macrophytes at the Powerline Milfoil sampling site from 1990 to 1992

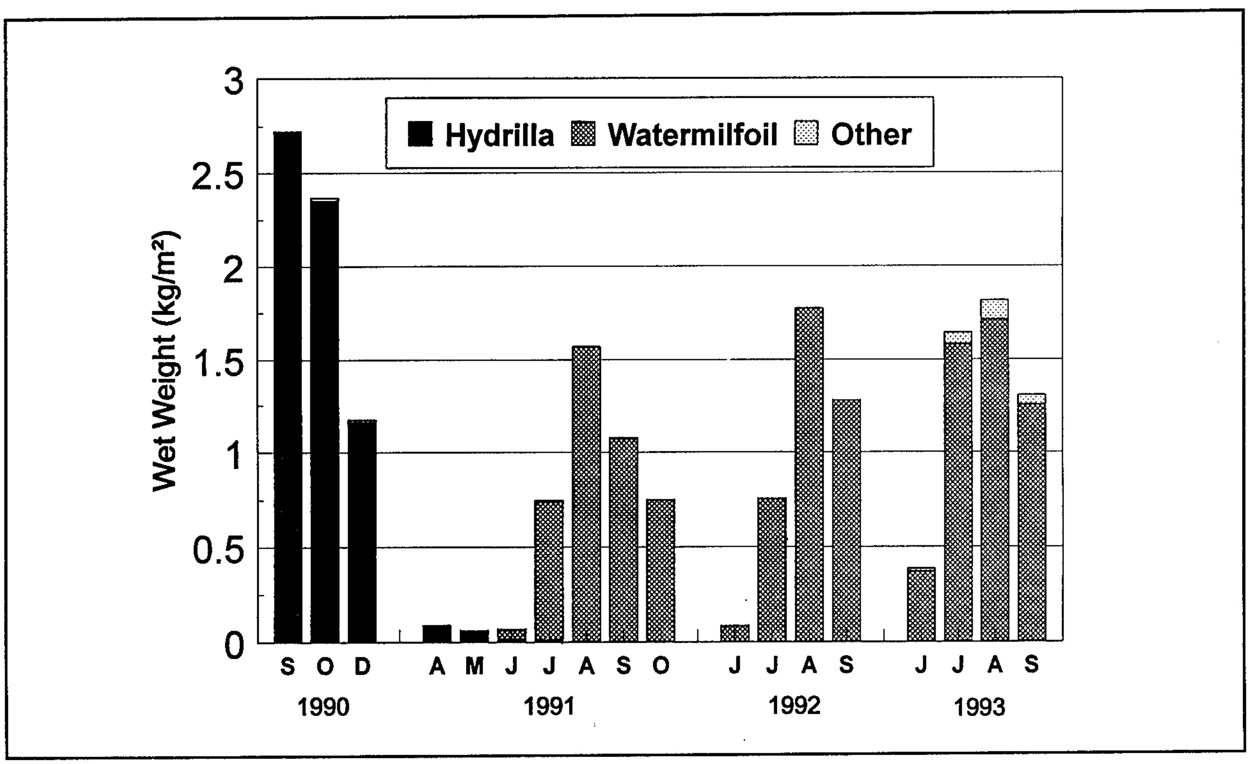

Figure 6. Mean aboveground biomass of aquatic macrophytes at the Brewster Hydrilla sampling site from 1990 to 1993

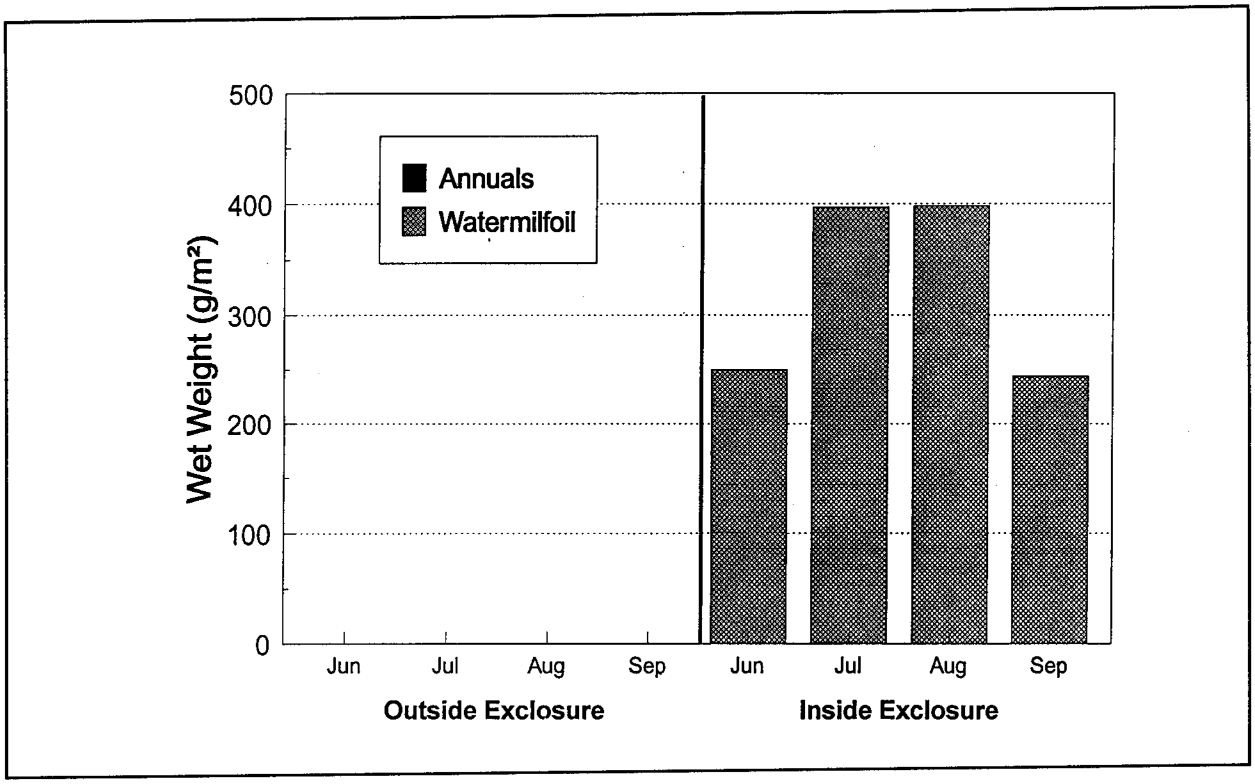

Figure 7. Mean aboveground biomass inside and outside of exclosure at the Brown's Creek sampling site in 1993

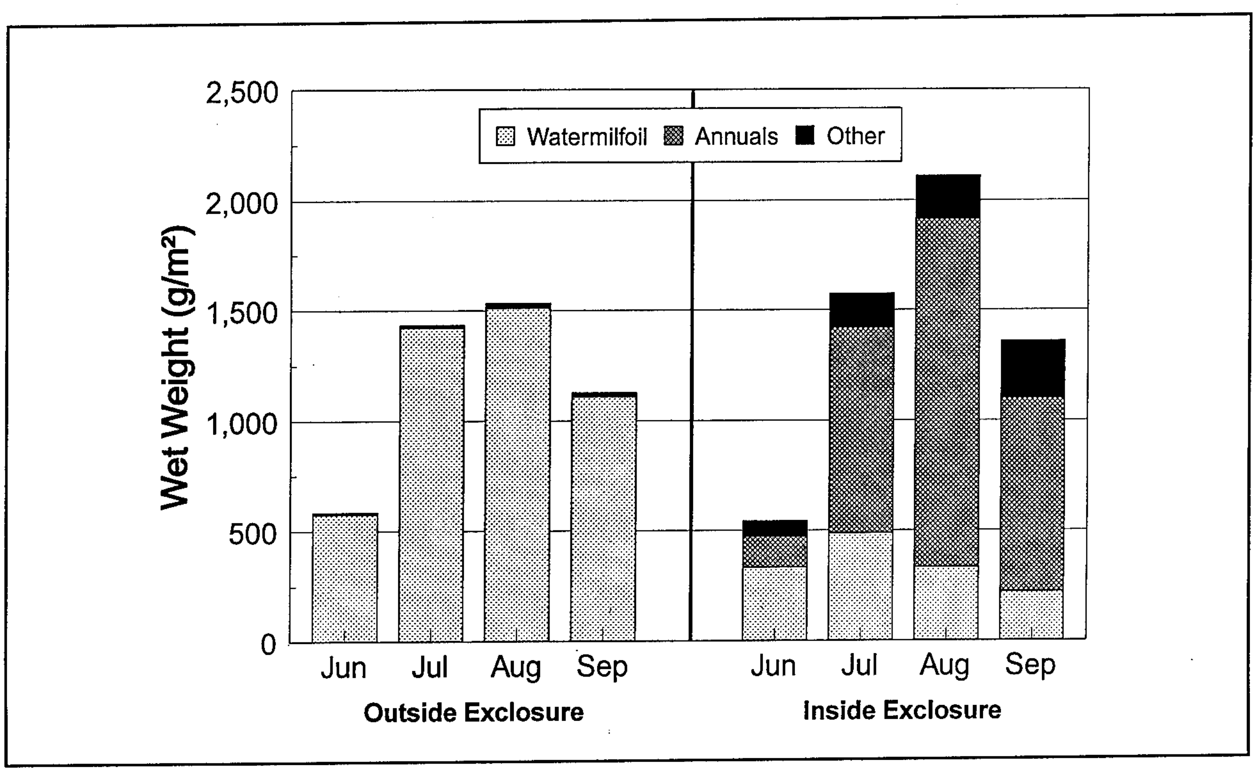

Figure 8. Mean aboveground biomass inside and outside of net exclosure at Chisenhall Shallow sampling site in 1993

habitat and coverage estimates from aerial photography have shown significant reductions in species such as spinyleaf naiad, southern naiad, muskgrass, narrow-leaved pondweeds, coontail, and hydrilla. Most of these plants rank high on the grass carp's food preference list (Miller and Decell 1984; Leslie et al. 1987), and grass carp are known to selectively feed on preferred plants before consuming less palatable species such as Eurasian watermilfoil. The abundance of spinyleaf naiad, muskgrass, and several other preferred species within exclosures constructed on Guntersville Reservoir in 1992 and 1993 supports selective feeding behavior by grass carp. In the first year of stocking (1990), grass carp were observed feeding in extremely shallow water of several embayments on Guntersville Reservoir. As the fish have increased in size, their reluctance to feed in these shallow areas may explain the observed increase in annual species in 1993 at scattered localities having slightly greater depths.

The extent to which grass carp were involved in the reduction of hydrilla from about 750 ha in 1989 to about 130 ha in the late summer of 1990 cannot be estimated from available data. Most of the 620 ha of hydrilla that declined was in relatively deep water (2 to 4 m) in the downstream portion of the reservoir that was more likely to have been adversely impacted by turbidity than hydrilla colonies from TRM 378 to TRM 384 that occurred primarily at depths of 1.5 to 2.0 m. However, grass carp are believed to have caused the reduction of 120 ha of hydrilla from TRM 378 to TRM 384 from 1990 to 1991. Large numbers of grass carp were observed feeding in "topped out" hydrilla colonies in the late summer and fall of 1990. The presence of hydrilla during the early summer of 1991 (Grodowitz and Snoddy 1992) in small exclosures constructed for insect biocontrol studies supports this conclusion. Underwater surveys indicated some regrowth of hydrilla in 1993 in a few areas of the reservoir that had abundant hydrilla in the late 1980s. However, hydrilla stems in some areas (e.g., Pine Island) appeared to have been cropped 0.1 to 0.3 m above the hydrasoil.

Even though Eurasian watermilfoil is not a preferred food species of grass carp (Miller and Decell 1984; Leslie et al. 1987), grass carp will consume it when more preferred species are unavailable (Leslie et al. 1987; TVA 1989). This may in part account for an almost total decline in 1991 of Eurasian watermilfoil in shallow water areas of Conner's Islands and Brown's Creek embayment in the downstream portion of Guntersville Reservoir where watermilfoil colonized about 240 ha in 1990. Regrowth of Eurasian watermilfoil in 1992 and 1993 in some downstream areas (TRM 363 to 373) of Guntersville may indicate a reduction in grass carp herbivory or improved growing conditions. Recolonization in some areas of the reservoir with sparse vegetation also may be slowed by other herbivores such as turtles.

Because it has been difficult to achieve intermediate vegetation levels using grass carp (Leslie et al. 1987), aquatic vegetation control with grass carp is often viewed as an "all or none" scenario with partial control being the exception rather than the rule. However, the "all or none" scenario may not be the case in large reservoirs (such as Guntersville) and in water bodies (TVA 1987, 1989) with a macrophyte community dominated by a species of low palatability such as Eurasian watermilfoil. In reservoirs such as Guntersville, hydrilla, naiads, and other highly palatable species may be significantly reduced without total loss or with minimal impact to nonpreferred plant species if grass carp are stocked at low to intermediate rates. The demonstrated feeding preference of grass carp for hydrilla (Miller and Decell 1984; Leslie et al. 1987) also potentially allows for significant reduction in this particularly invasive species that can rapidly colonize the littoral zone of a reservoir such as Guntersville to depths of 5 m. Stocking of grass carp while hydrilla is in the early stages of establishment within a water body and has a restricted distribution should minimize impacts to nontarget species.

Even when using conservative stocking rates in large systems dominated by nonpreferred species, there are uncertainties and

some risks. Feeding may occur in an area where a diversity of aquatic macrophytes is desirable (e.g., wildlife management areas), and the grass carp may move out of the portion of a water body or emigrate to other waters in open systems. Even when grass carp are stocked at conservative rates, natural declines in macrophyte populations may result in more than anticipated, or desired reductions in aquatic vegetation. However, in spite of these uncertainties, the potential benefits of suppressing the spread of species such as hydrilla may fully warrant grass carp use in multipurpose, open-reservoir systems in some instances.

Acknowledgments

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Grass Carp Collection, Aging, and Growth in Large Water Bodies—A Status Report for 1992-1993

by

James P. Kirk,[1] *James V. Morrow, Jr.,*[1] *and K. Jack Killgore*[1]

Introduction

In recent years, triploid and diploid grass carp (*Ctenopharyngodon idella*) have been stocked in reservoirs and lakes to control nuisance aquatic vegetation such as hydrilla (*Hydrilla verticillata*). The U.S. Army Engineer Waterways Experiment Station (WES) developed a grass carp stocking model (Boyd and Stewart 1992), but need objective data to refine the model for different regions and types of vegetation. This report summarizes efforts during 1992 and 1993 to obtain basic biological information on grass carp that will be used to refine the stocking model.

Our objectives were to develop cost-efficient collection techniques, length-to-weight relationships necessary for use in backcalculation, and methods to age the fish. As these techniques are developed and information gathered, estimates of growth, mortality, and standing stocks of grass carp in large water bodies will be made available to refine the stocking model.

Methods

Grass carp were collected from two major reservoir systems, Lake Guntersville, Alabama, and the Santee Cooper reservoirs (Lakes Marion and Moultrie) in South Carolina. Lake Guntersville is a 68,000-acre reservoir in northern Alabama managed by the Tennessee Valley Authority. A total of 118,400 diploid and triploid grass carp were stocked between 1988 to 1990 to control hydrilla.[2] The Santee Cooper lakes total approximately 170,000 acres and have a major hydrilla infestation. Lake Marion was stocked with 100,000 triploid grass carp per year from 1989 through 1992 to control hydrilla. An additional 50,000 triploid grass carp were stocked into Lake Moultrie during 1993 to control spreading colonies of hydrilla.

An initial problem was collecting adequate numbers of triploid grass carp using traditional sampling gears. As part of a grass carp monitoring program (Morgan and Killgore 1990), vegetated areas of Lake Marion were extensively electrofished. In addition, the South Carolina Wildlife and Marine Resources Department (SCWMRD) regularly conducted gill net, cove rotenone, and electrofishing sampling of the Santee Cooper system. Although some grass carp were collected by each of these methods, the numbers collected were inadequate for research needs.

Triploid grass carp were accidentally harvested during bowfishing tournaments in 1990. As a result, bowfishermen were used in a final attempt to collect triploid grass carp. The collection was organized and permitted by SCWMRD. Bowfishing teams familiar with the reservoirs and who had consistently won local bowfishing tournaments were recruited and paid for their efforts.

The collection efforts took place at night using a boat with five lights powered by a small gasoline-powered generator. Bowfishermen stood on the bow of the boat and maneuvered into coves and along shallow flats looking for grass carp. Fish were generally collected in coves and other confined areas as they swam by the boat. Fish had to be quickly identified and shot in depths up to 6 ft. Once the fish

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[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.
[2] Personal Communication, 1994, David Webb, Tennessee Valley Authority, Muscle Shoals, AL.

was shot with a conventional bowfishing arrow, it was played on a line using a reel attached to the front of the bow. After the fish tired, it was reeled near the boat, shot a second time to prevent escape, and then gaffed. Collected fish were processed the next day. Biological information, to include a survey of stomach contents, was collected by biologists.

Length-to-weight relationships are used to indicate the condition or "plumpness" of a fish population (Le Cren 1951). Another use is to predict weights of fish of a given length determined by backcalculations using scales or otoliths. Total lengths to the nearest millimeter and weights to the nearest gram were measured for each fish collected. The length-to-weight relationship was computed using a power function (Ricker 1975):

$$\text{Weight} = \text{intercept} \times \text{Length}^{\text{slope}}$$

Generally, fish are aged by examining annual marks on scales, otoliths (ear bones), or other bony structures (Jearld 1983). However, we did not find any information on grass carp aging techniques and little information on growth rates. Consequently, we evaluated the potential of otoliths and scales as aging structures to develop grass carp growth and mortality information.

Age and growth information presented in this paper was obtained by examining scales; we did, however, compare ages determined from otoliths and scales from grass carp collected in Lake Guntersville. Scales had consistently recognizable annuli and were examined by projecting their image on a microfiche projector. Distances to each annuli and distance to the margin of the projected image were measured using a GP-7 graphbar digitizer and a personal computer. The computer is equipped with a series of basic programs used to measure structures and backcalculate lengths of fish. Backcalculated lengths were estimated using the Fraser-Lee Method (Carlander 1982) which uses the following formula:

$$Li = a + (Lc\text{-}a/Sc)Si$$

Where

Li = calculated length at age i
a = intercept of the body scale regression
Sc = diameter or radius of scale at capture
Lc = length of the fish at capture
Si = scale measurement at annulus i

Results

Electrofishing, netting, and rotenone were unsuccessful in collecting sufficient numbers of grass carp (Kirk et al. 1993). Skilled bowfishermen proved an effective, cost-efficient method to collect grass carp. A total of 69 and 125 triploid grass carp were collected in Lakes Moultrie and Marion in 1992 and 1993, respectively. Substantially less effort was expended on Lake Guntersville where tournament bowfishermen were not recruited in time to make a concerted collection effort. However, 43 grass carp were shot during a bowfishing tournament in April 1993, and these fish were used to provide initial estimates of age and growth.

The length-to-weight relationship for grass carp from Santee Cooper collected during 1993 has not yet been determined. The length-to-weight relationship for fish collected on Lake Guntersville follows:

$$\text{Weight} = 0.0000045\ \text{Length}^{3.163}$$

The relationship for Santee Cooper triploid grass carp collected in 1992 follows:

$$\text{Weight} = 0.0000027\ \text{Length}^{3.25}$$

After calculating the length-to-weight relationship, we backcalculated lengths and weights for grass carp from Lake Guntersville using scales and otoliths from the 43 fish. Table 1 compares age-specific sizes for the Santee Cooper reservoirs and Lake Guntersville for grass carp collected during 1992 and 1993. Grass carp in the Santee Cooper reservoir are growing more rapidly.

Table 1
Age-Specific Lengths and Weights of Grass Carp from Lake Guntersville, Alabama, and the Santee Cooper Reservoirs, South Carolina

	Lake Guntersville		Santee Cooper	
	Length mm	Weight g	Length mm	Weight g
Age I	311	339	361	547
Age II	596	2,699	698	4,894
Age III	746	5,491	821	8,294
Age IV	845	8,144	908	11,506
Age V	899	9,907	n/a	n/a

Sagittal otoliths of cyprinids often show no clear marks and are not suitable for age determination. When aging cyprinids, the lapillus (utricular otoliths) should be used (Victor and Brothers 1982). The lapillus appears to lay down annuli and should be a suitable aging structure to compare with scales. Figures 1 and 2 show all three otoliths and a sectioned lapillus, respectively. Almost complete agreement existed in ages determined from sectioned otoliths and scales from fish collected from Lake Guntersville.

Discussion

Skilled bowfishermen are an effective method of collecting grass carp in large water bodies. We intend to expand our collection efforts and increase sample sizes in both systems in 1994. Instead of relying on grass carp collected at Lake Guntersville bowfishing tournaments, we will attempt to contract with individual bowfishermen and collect between 100 and 200 fish during 1994.

Initial estimates of growth strongly suggest that grass carp in the Santee Cooper reservoirs are growing at a much faster rate than in Lake Guntersville. This may be related to the availability of preferred foods. Some fish collected from Lake Guntersville in early April 1993 were eating filamentous algae and other less preferred foods (Sutton and Vandiver 1986; Leslie et al. 1987) Hydrilla, a preferred food of grass carp, is almost entirely gone from Lake Guntersville, while extensive stands of Eurasian watermilfoil (*Myriophyllum spicatum*), a less preferred food, is not being consumed by grass carp. Hydrilla in the Santee Cooper system is being controlled in upper Lake Marion, but is rapidly expanding into other parts of the system, especially Lake Moultrie. Thus, a preferred food is not limiting in this system.

Preliminary results suggest that scales and otoliths are suitable for aging grass carp, and research is ongoing to validate these aging structures. There was high agreement between sectioned utricular otoliths and scales; the backcalculated length of Age I fish was very close to the length of known Age I fish stocked into the Santee Cooper reservoirs and Lake Guntersville.

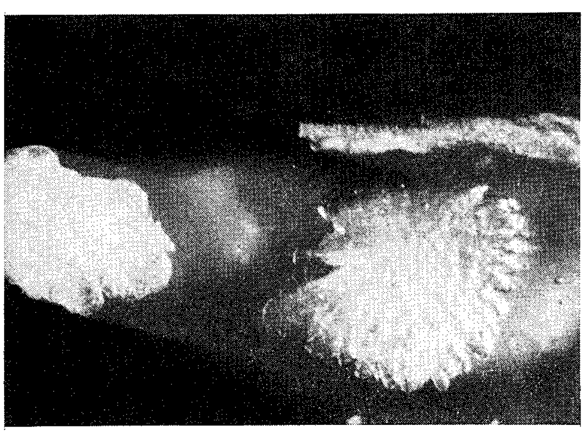

Figure 1. A view of grass carp otoliths. Left most is lapillus, right top is asteriscus, and right bottom is sagittus.

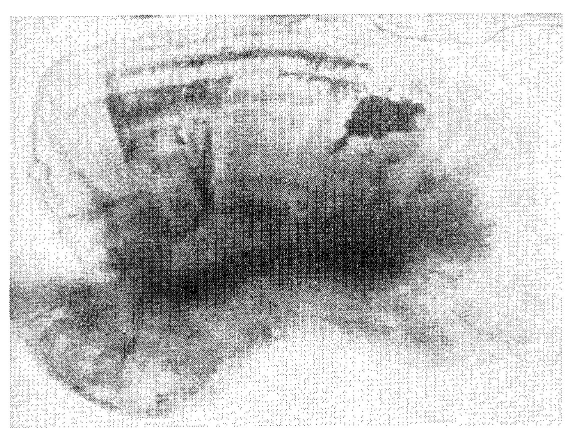

Figure 2. Sectioned utricular otolith (the lapillus), showing annuli

Future efforts will focus in several directions. Scales and otoliths require validation. Fish collected from the Santee Cooper system in 1993 and fish to be collected from both systems in 1994 must be aged to generate growth and mortality estimates. This information can be incorporated into a Ricker Table (Ricker 1975) to provide estimates of numbers, biomass of grass carp by age class, and total grass carp biomass. The primary relationship used to derive these estimates follows:

$$N_t = N_0 e^{-zt}$$

This relationship states the number at time (t) is equal to the initial number (0) raised to the instantaneous rate of total mortality (z) times the number of years (t) (Ricker 1975).

Triploid grass carp have potential to control aquatic vegetation in large water bodies (Allen and Wattendorf 1987). However, their use must not result in overstocking or understocking (Kirk 1992). Basic biological data are needed to refine parameters used in the stocking model in order to achieve appropriate stocking densities. Our future estimates of mortality, growth, and biomass should improve stocking strategies and allow managers to more accurately predict reductions in aquatic vegetation.

Acknowledgments

We thank Dr. John Mark Dean of the University of South Carolina's Baruch Institute for his assistance in locating otoliths and recommending suitable aging structures. Val Nash, Miller White, and Scott Lamprecht of the South Carolina Wildlife and Marine Resources Department provided invaluable assistance in collecting and processing triploid grass carp from the Santee Cooper reservoirs.

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Grass Carp Herbivory at Guntersville Reservoir and Its Impact on Waterfowl—Preliminary Results

by
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Grass carp herbivory of Eurasian watermilfoil and native aquatic macrophytes was examined at North Sauty Creek, Guntersville Reservoir, Alabama, in the summers of 1992 and 1993, by comparing biomass of exclosed and grazed samples. In both years, grass carp herbivory had no significant impact on milfoil biomass ($P < 0.05$), but had a significant negative impact on native plant biomass ($P < 0.05$). We compared waterfowl use of milfoil and native plant (i.e., *Chara* spp. and *Najas* spp.) areas during October 1993-February 1994 to determine the impact of grass carp herbivory on migrating and wintering waterfowl. We examined natural senescence of milfoil and native plants using 1-m^2 exclosures to track availability of the two vegetation types. We also quantified waterfowl herbivory and use of milfoil and native plant areas. Preliminary results show that while milfoil persisted into February, native plants senesced much earlier, in December. Waterfowl herbivory was significant ($P < 0.05$) in both milfoil and native plant areas in all months except October, where only herbivory of natives was significant. Waterfowl use of milfoil remained constant throughout the fall and winter. However, native plants attracted a greater number and diversity of ducks than milfoil in October and November.

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Grass Carp in Guntersville Reservoir: A Review of Fisheries Impact Assessments

by

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Field evaluation of the impact of grass carp on the fish fauna in Guntersville Reservoir in Alabama now spans more than a decade. Assessments began with the trial release of 4,500 monosex fish in a 160-ha embayment and are continuing with the introduction of 100,000 triploid fish in 1990 in the open reservoir, a 27,500-ha impoundment of the Tennessee River. The initial phase (1983-1988) was oriented more to the total fish community (Tennessee Valley Authority (TVA) 1987, 1989), while the current emphasis has been on the sport fishery. However, standing stock estimates of largemouth bass from the first phase served as an important reference. The largemouth bass fishery in this eutrophic reservoir is generally considered outstanding, with the weight of the catch for a 3-day tournament (300 to 400 anglers) often exceeding 3,000 kg. A substantial fishery also exists for the following: bluegill and redear sunfish, black and white crappie, white bass, blue and channel catfish, and sauger. In this review, some of the key results to date are discussed relative to determining or predicting an impact of grass carp on the sport fishery in Guntersville Reservoir.

Bain (1993) has provided an independent perspective on the utilization of grass carp in this reservoir, leading up to the 1990 stocking. He used Guntersville as a case study in developing a checklist for assessing impacts of introduced species, particularly grass carp in large systems. The list was grouped into six categories: population control, hybridization, diseases and parasites, habitat alterations, biological effects, and management issues. Following his protocol, results in this review primarily address potential impacts related to population control (grass carp reproduction and recruitment) and management issues (sport fishery).

Also, the TVA assessment of grass carp in this reservoir was incorporated into the Joint Agency Guntersville Project (JAGP) in 1990. This 5-year demonstration, sponsored by TVA and the U.S. Army Corps of Engineers (USACE), was established to assess a broad array of issues pertaining to aquatic plant management, including the effectiveness and impact of various plant control methods in addition to grass carp. In this context, we judged that the common variable or agent for potential fisheries impacts is the reduction of aquatic plant biomass or areal coverage. The effects of changes in areal surface coverage, expressed as a percentage of total reservoir area, are examined here.

Stocking Rate and Aquatic Plant Conditions

Between late April and mid-July in 1990, a total of 100,000 grass carp were stocked at 23 locations in the lower two-thirds of Guntersville Reservoir. The objective was to stock grass carp at a low-to-intermediate rate to substantially reduce hydrilla and obtain only limited control of other taxa, particularly Eurasian watermilfoil (TVA 1990). As this plan was developed in 1988, the surface coverage of aquatic macrophytes exceeded 8,000 ha, or about 30 percent of the reservoir. Following approval by the Alabama Game and Fish Department to stock 12 triploid grass carp per vegetated hectare, contracts were initiated in spring 1989 to obtain fish for stocking in

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1990. However, because of a natural decline of vegetation that began in summer 1989, the stocking rate was 17/hectare of vegetation by the time of stocking in 1990. The actual density of grass carp in Guntersville Reservoir was further confounded by the legal release of about 20,000 grass carp during 1987-1989 by private citizens or lake user groups. Under these circumstances, however, the estimated density of grass carp following stocking in 1990 did not exceed 20 per vegetated hectare. In comparison, grass carp were stocked at a rate of 47/hectare of vegetation in the trial study, which resulted in no adverse fisheries impacts.

Areal surface coverage of aquatic vegetation was determined from an aerial photographic survey conducted each September. As indicated, aquatic macrophytes in this reservoir were apparently undergoing a natural decline as the grass carp were introduced. However, for additional analysis of this issue and a more complete description of aquatic plant conditions related to grass carp in Guntersville Reservoir (see Webb et al., this volume). As a reference for this fisheries review, the levels of percent coverage since 1988 were as follows:

1988	1989	1990	1991	1992	1993
29	21	12	8	9	11

This unexpected and rapid decline in plant coverage compromised fisheries assessments relative to intermediate and high levels of plant coverage, as the original minimum target level was 10 percent.

Population Control

Normally, the use of triploid grass carp would preclude the need for monitoring for reproduction and recruitment (Bain 1993). Also, successful spawning by diploid fish appeared to be a remote possibility, based on the literature and the absence of grass carp larvae in numerous larval fish samples collected in this reservoir for assessing power plant siting and operations. Conditions for grass carp spawning include the following: water temperature, 23 to 28 °C; high turbidity; and sustained flows of 0.6 to 1.8 m/s for a distance of 40 to 50 km (Stanley, Miley, and Sutton 1978). Water temperatures in this range do not occur until late May in this reservoir, when the probability of required flows is greatly diminished, occurring only in a rare flood event. However, given the presence of diploid fish in the reservoir from previous introductions, monitoring for evidence of grass carp reproduction was conducted to allay public concern on this issue and for experimental control.

Because grass carp larvae typically move into vegetated overbank areas a few days after hatching, a sampling regime utilizing light traps was selected as the best monitoring method for early detection. The light traps used were a modification of the quatrefoil-design (Floyd, Courtenary, and Hoyt 1984). Four transects in the middle-to-lower portion of the reservoir were sampled for 6 weeks beginning in June. Seven traps per transect were set after dark and fished for 2 hr. Samples were preserved and processed in the laboratory. Although large numbers of larval and juvenile fish were collected, including several taxa, no grass carp larvae were collected in 1990-1993. Also, no juvenile grass carp were collected in cove rotenone surveys conducted at five sites in mid-July during this time period.

The other issue relative to population control is the dispersal of grass carp from Guntersville Reservoir that could result in concentrations of fish with undesirable effects in other reservoirs upstream or downstream. To date, we have no evidence that this has occurred.

Sport Fishery

A comprehensive sportfishing census or creel survey, based on nonuniform probability sampling, was initiated in February 1990, to estimate total recreational angler effort, fish harvest, catch rate, and trip expenditures (Fisheries Information Management Systems,

Auburn, AL). The survey was stratified spatially and temporally. Temporal stratification included 10 monthly blocks (February-November) divided into weekdays (Monday-Thursday) and weekend days (Friday-Sunday). The reservoir area sampled was divided into three strata, each approximately 8,000 ha, designated as lower, middle, and upper reservoir. Sampling effort for each of two roving clerks was 20 days per month.

Total angler effort (nontournament fishing) was highest in 1990, 1.6 million hr, but declined 63 percent by 1993. The decline in fishing effort occurred among both boat and bank anglers (Figure 1). Likewise, total number of fish caught peaked in 1990, >2 million, and declined 70 percent by 1993. Except for largemouth bass, catch rates remained stable or increased (Figure 2). Angler effort and total catch were consistent among strata over time: highest values in the lower zone (A), followed by the middle (B), and then the upper zone (C). However, the levels of plant coverage in these three zones were just the opposite: 24 to 30 percent (C), 6 to 9 percent (B), and 0.2 to 4 percent (A). More than 90 percent of the effort was by Alabama resident anglers, with an equal percentage of anglers traveling less than 160 km. Economic value, based on trip expenditures plus the willingness-to-pay value, ranged from $4.15 million in 1990 to $1.65 million in 1993, although the hourly rate was higher in 1993 ($2.75/hr) than in 1990 ($2.57/hr). Trip expenditures alone declined $1.4 million during this period, with the largest decline ($0.8 million) occurring between 1992 and 1993.

Targeted angling effort for largemouth bass comprised two-thirds of the total angler effort in this reservoir throughout 1990 to 1993 (Figure 3). Approaching 1.0 million hr in 1990, it declined 58 percent by 1993. Catch rate (release and harvest) ranged from 0.66/hr in 1990 to 0.44/hr in 1993, which was a slight increase to 1992. Biomass harvested (removed from the reservoir) was 145,000 kg in 1990, but it declined more than 80 percent by 1993 because of the reduction in effort and a 66-percent decrease in the harvest rate of bass. However, mean weight of bass harvested

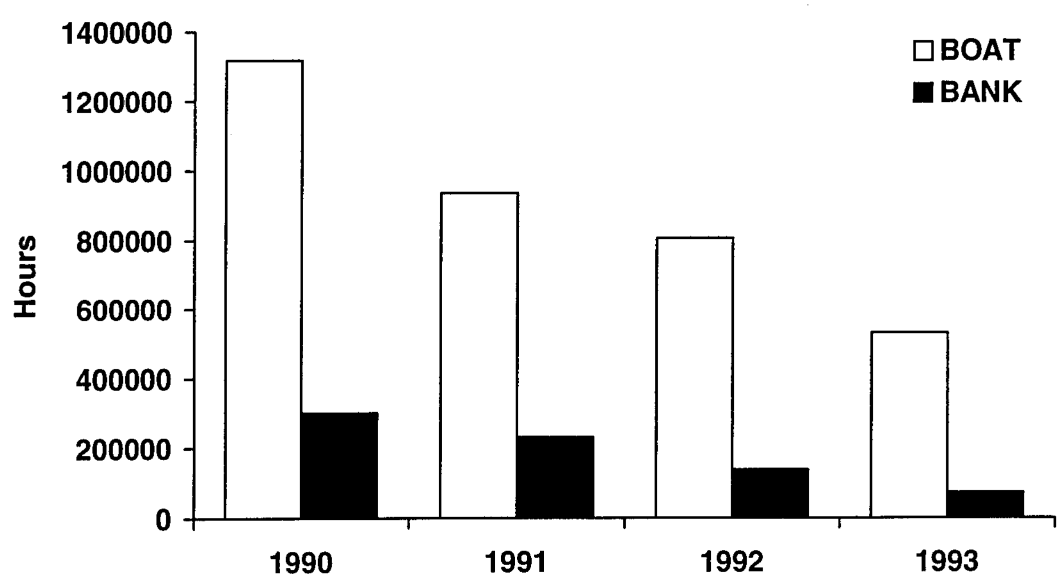

Figure 1. Total effort (hours) of boat and bank anglers during 1990-1993, Guntersville Reservoir

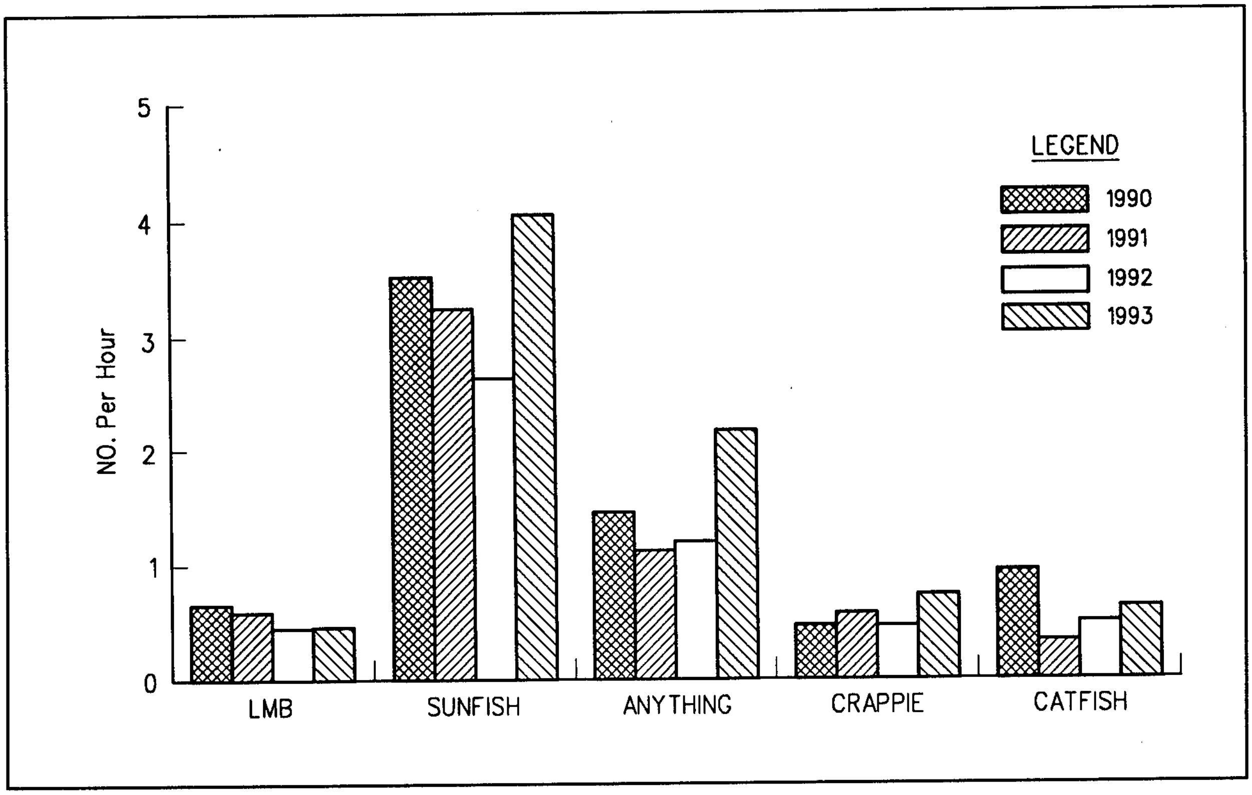

Figure 2. Catch rate (release and harvest) of five categories of sportfish during 1990-1993, Guntersville Reservoir (LMB–Largemouth Bass)

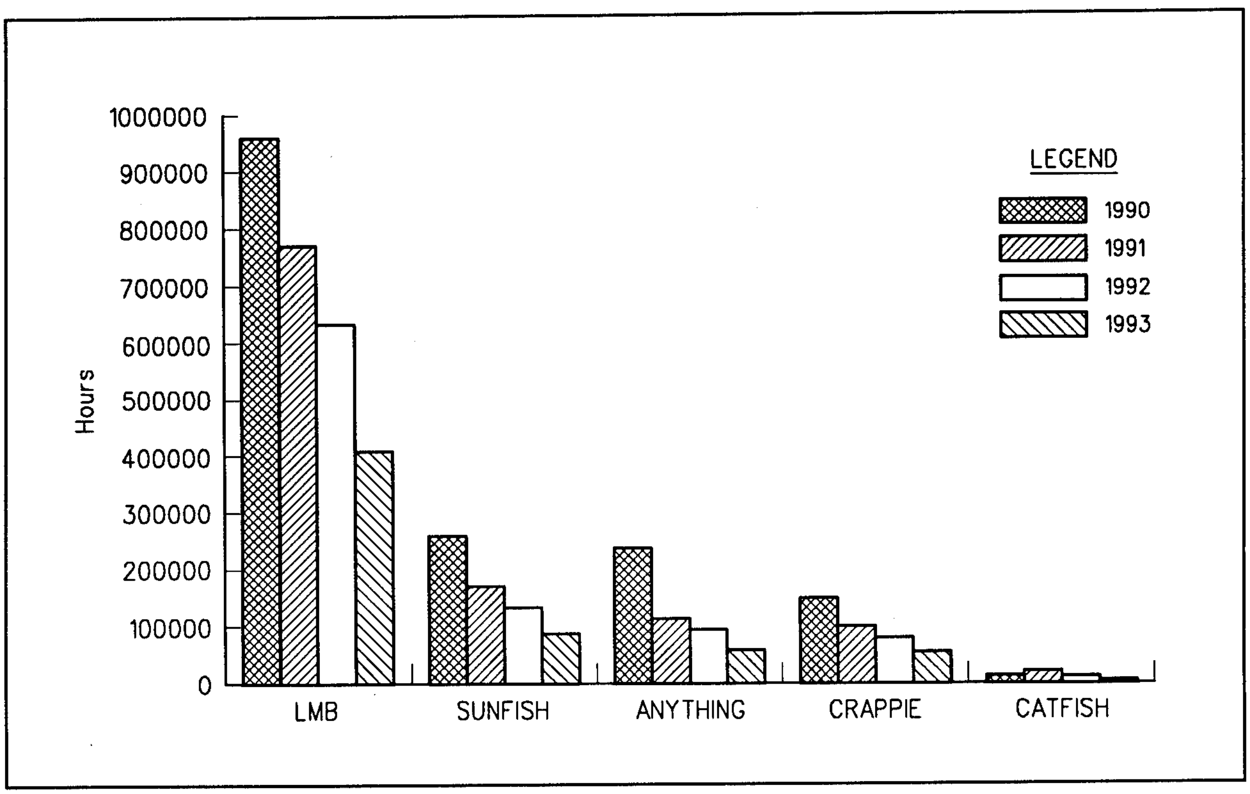

Figure 3. Angler effort (hours) expended for five categories of sportfish during 1990-1993, Guntersville Reservoir (LMB–Largemouth Bass)

was higher in 1993 (0.79 kg) than in 1990 (0.62 kg). The release rate of bass was relatively stable, 0.48/hr in 1990 to 0.38/hr in 1993, while the harvest rate declined from 0.18/hr to 0.06/hr. Partly because of the large numbers of bass harvested in 1990-1991, variable year-class production, and faster growth rates, a 381-mm minimum length limit was imposed on this reservoir in October 1993. Prior to this, there was no length limit; so the release rate was voluntary.

Largemouth Bass Population Status

To determine the standing stock and relative abundance of largemouth bass in Guntersville Reservoir, cove rotenone surveys and electrofishing were selected initially as reliable methods to monitor the bass population as well as various other species in the fish community. Rotenone samples were collected at five coves (1.2 ha) in mid-July in 1990-1993, and electrofishing samples (10 timed-runs) at nine stations were collected in September-October. Although autumn electrofishing was considered to provide a consistent estimate of relative abundance and an adequate sample of bass to examine size structure of the population, density and biomass estimates of harvestable-size (>250-mm total length (TL)) largemouth bass from standard 1.2-ha rotenone surveys were questionable. Results of the creel survey in 1990-1991 showed a mean harvest rate for bass of 5.2 kg/ha, whereas the mean standing biomass estimate for this period was 6.8 kg/ha, suggesting that fishing mortality was extremely high or the biomass estimate was low. Based on the continuing high catch rate in the creel census and limited spring electrofishing samples, it appeared to be the latter. In spring 1992, an electrofishing catch-depletion technique was evaluated (Maceina, Wrenn, and Lowery 1994) and subsequently included in the sampling regime for spring 1993 and 1994. Basically, this entailed block-netting seven coves (total area 21.5 ha) and depleting the bass population by multiple electrofishing runs. This provided another estimate of standing biomass and a large sample of bass for size structure and age-growth determinations.

Although comparable creel and electrofishing surveys were not conducted during the 1980s when aquatic plant coverage was the highest, (23 to 29 percent), large scale-rotenone surveys (11 ha) were conducted (1983, 1986, and 1988) within the 160-ha trial embayment in which the level of macrophyte coverage was as high as 75 percent. Biomass estimates of harvestable largemouth bass from these surveys ranged from 36 kg/ha in 1983 to 30 kg/ha in 1988. For comparison, biomass estimates determined by electrofishing catch-depletion in 1993 and 1994 were 33 and 32 kg/ha, respectively (Table 1). The accuracy of both sampling methods (large rotenone and catch-depletion electrofishing) for determining the true population biomass or density of largemouth bass in this reservoir will always be uncertain. However, in our opinion, it is unlikely that actual biomass in the 1980s was significantly higher than the estimates reported here. Under similar conditions (higher water clarity, increasing aquatic macrophytes, and longer water retention time) in 1986 in Kentucky Reservoir (350 km downstream), the standing biomass of largemouth bass was 21 kg/ha (Buynak, McLemore, and Mitchell 1991).

Table 1
Biomass of Harvestable (>250-mm TL) Largemouth Bass and Population Size Structure in Guntersville Reservoir, 1983-1994

Year	Method[1]	Biomass kg/ha	Relative Stock Density[2] PSD (40-70)	RSD-P (10-40)	RSD-M (0-10)
1983	LR	36	40	8	1
1986	LR	31	30	7	0
1988	LR	30	16	7	1
1990	AF	–	48	21	7
1991	AF	–	43	18	4
1992	AF	–	53	25	8
1993	CD	33	63	26	7
1994	CD	32	46	17	5

[1] Sampling method: LR –large rotenone; AF–autumn electrofishing; CD–catch-depletion electrofishing.
[2] Relative Stock Density (Gabelhouse 1984), PSD = 100 (quality-length fish/stock-length fish); RSD-P = percentage of bass >381-mm TL; RSD-M = percentage of bass >508-mm TL.

Both methods did provide large samples of bass for examining population size structure under disparate levels of aquatic plant coverage. Based on Relative Stock Density (RSD) ratios (Gabelhouse 1984), the current size structure of the largemouth bass population in this reservoir is balanced, whereas the structure in the 1980s reflected an imbalance toward smaller fish (<304-mm TL). Accordingly, the RSD-P ratio (percentage of bass with TL equal to or greater than 381 mm) in the 1980s was below the accepted range, 10 to 40 percent (Table 1). The decline in PSD and RSD-P ratios in 1994 occurred primarily because of highly successful recruitment of the 1992 year class to stock-size (203- to 303-mm TL), while growth rates for age-1 and age-2 bass increased from 1993 (exceeding Alabama statewide mean growth rate).

Additional sampling (small-plot rotenone and electrofishing surveys) during 1990-1993 was conducted specifically to assess reproductive success and early recruitment of largemouth bass (Maceina, Rider, and Lowery 1993). Given the large numbers of young bass that are commonly found in aquatic plant colonies in this reservoir and in other water bodies (Durocher, Provine, and Kraai 1984; Moxely and Langford 1985), we assumed a strong link between aquatic plant abundance and recruitment of age-1 bass to the population. However, as noted in the following summary, this was not necessarily a valid assumption.

From 1990 to 1993, age-0 largemouth density in August was higher in submersed vegetation than unvegetated areas for 3 of these 4 years (Table 2). However, even though age-0 abundance of bass was generally greater in vegetated regions in late summer, electrofishing catch rates of these fish at age-1 indicated abundance in upstream vegetated areas was higher for only one of these 4-year classes produced between 1990 and 1993. In fact, abundance of age-1 bass from the 1992 year class produced in vegetation was less than that estimated in unvegetated regions in June 1993 (Table 2). This occurred even though density was about nine times more abundant in vegetation when estimated in August 1992 when these fish were age-0.

Table 2
Density and Relative Abundance of Age-0 and Age-1 Largemouth Bass in Guntersville Reservoir, 1990-1993

Year Class	Collection Time	Variable	Habitat: Plant	Habitat: No Plants
1990	Aug 90	N/hectare	48	71
	Nov 91	N/hour	16	20
1991	Aug 91	N/hectare	387	22*
	Apr 92	N/hour	14	6*
1992	Aug 92	N/hectare	883	104*
	Mar 93	N/hour	9	8
	Jun 93	N/hour	14	31*
1993	Aug 93	N/hectare	366	104*
	Mar 94	N/hour	28	38

Note: *Significant ($P < 0.05$) difference between values in rows.

Young largemouth bass inhabiting aquatic vegetation in Guntersville Reservoir usually grew slower than those fish collected from unvegetated areas (Table 3). Size differences between vegetated and unvegetated habitats were not due to differences in spawning time as indicated by daily growth rings. Back-calculated growth rates of young largemouth bass as young as 30 days old were less in vegetated than in unvegetated areas, and by August, daily growth rates were about 10 to 20 percent faster in unvegetated habitats. Slower growth conferred smaller size by the fall of each year, and size-dependent mortality likely caused a reduction of reproductive success by age-1 and contradicted what seemed apparent by age-0. Even though greater size-dependent mortality was evident for young bass inhabiting vegetated areas, size at age-1 was generally less than observed for fish inhabiting unvegetated regions (Table 3).

Trophic production (directly or indirectly via algal biomass or zooplankton) did not appear related to differences in age-0 bass density and growth. In spring-early summer 1991, 1992, and 1993, crustacean macrozooplankton densities were either similar or higher in unvegetated than in vegetated habitats. Thus, aquatic vegetation provided habitat and environmental conditions conducive to high abundance of age-0 bass. Likely, mechanisms for greater age-0 abundance in vegetation included a reduction in predation, given that

Table 3
Mean Total Length (mm) of Age-0 and Age-1 Largemouth Bass in Guntersville Reservoir, 1990-1993

		Habitat	
Year Class	Collection Time	Plant	No Plants
1990	Aug 90	83	98*
	Nov 91	244	236
1991	Aug 91	70	92*
	Nov 91	123	145*
	Apr 92	150	151
	Aug 92	196	229*
1992	Aug 92	64	72*
	Oct 92	74	132*
	Mar 93	144	183*
	Jun 93	185	184
1993	Aug 93	69	97*
	Oct 93	137	142
	Mar 94	176	203*

Note: *Significant (P < 0.05) difference between values in rows.

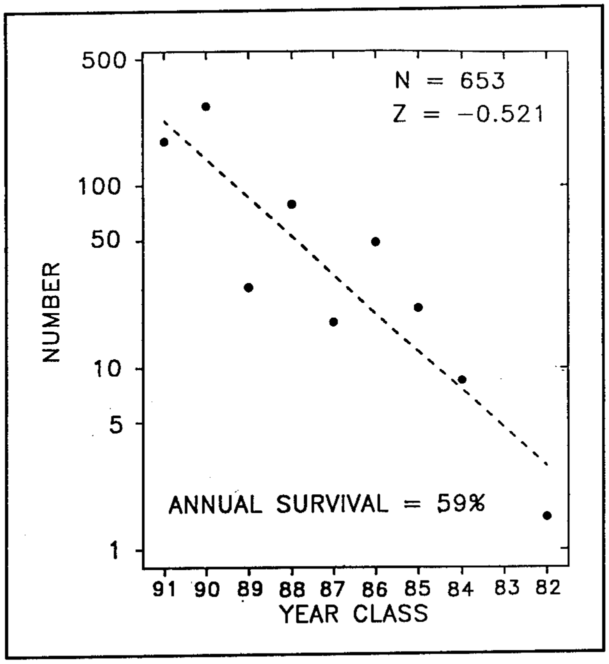

Figure 4. Number of age-2 (1991 year-class) to age-11 (1982 year-class) largemouth bass plotted against year class. Z represents the slope coefficient of the regression (dashed line) and is the instantaneous mortality rate

plants provided refuge from predation, and dense populations of macroinvertebrates, other than macrozooplankton, provided food (Watkins, Shireman, and Haller 1983; Schramm and Jirka 1989).

Catch-curve analyses (Ricker 1975) of fish age-2 and older conducted in 1993 show alternating year-class strength in this reservoir (Figure 4). Variable year-class strength was not related to plant coverage, but to retention time in April, which coincided with the initiation of largemouth bass spawning. For example, strong year classes were produced in 1986, 1988, and 1990 when late summer plant coverage was 24, 29, and 12 percent, respectively. However, weak year classes were produced in 1987, 1989, and 1991 when plant coverage was high at 23 and 21 percent, but then declined to 8 percent in 1991. The strong year classes of 1986, 1988, and 1990 were associated with April retention times 68, 43, and 25 days, while April retention times for the weak year classes of 1987, 1989, and 1991 were 18, 17, and 11 days, respectively.

Thus, when other environmental conditions are favorable, high levels of aquatic plants may not greatly enhance largemouth bass recruitment in this reservoir. However, when reproductive success is low in unvegetated habitats, aquatic plants appeared in some instances to augment overall recruitment to the population. This occurred in 1991 when age-0 bass density was 18 times less in unvegetated areas and catch rates of this year class at age-1 were twice as high in vegetation the following spring. In support of this, age-2 fish from the 1991 year class collected in 1993 were 30 percent more abundant in upstream vegetated than in downstream unvegetated habitats.

Conclusions

Based on the results examined through 1993, the impact of grass carp on game fish populations in Guntersville Reservoir has been minimal. The status of the largemouth bass population, which receives two-thirds of the total fishing pressure, is good to excellent. Size structure of the population is balanced, current growth rates are high, and the standing biomass is substantial. All of the above are important, particularly in relation to the current fisheries management objectives for this reservoir, which include a 381-mm minimum length limit. In fact, one point of concern is

the stocking-piling of fish below the legal limit. This condition can be accelerated or compounded by depressed growth rates of younger bass (<age-4). Negative effects of bass density and excessive aquatic plant structure can, in turn or combination, reduce growth rates substantially. This phenomenon has been demonstrated in this reservoir and in other studies (Colle and Shireman 1980; Bettoli et al. 1992).

Although the presence of grass carp or the reduction of plant coverage has had little or no impact on game/fish populations, these factors appeared to impact on the quality of the sport fishery and the level of angler effort in this reservoir. Both of these parameters, to a great extent, are determined by the angling public. Also, the interaction of quality and level of effort cannot be discounted. Considered separately, the 63-percent decline in total angler effort since 1990 is definitely an area of concern because of the probable decline in revenue to the local economy and possible impact on fishing license sales. We did not analyze or determine the multiple variables that can affect angler effort or the definition of quality. However, for the purpose of this discussion, the following conditions could be considered. For anglers that routinely catch bass at a rate of 0.66/hr, the current (1993) catch rate of 0.44/hr would be a decline in quality. However, catch rates from other established largemouth bass fisheries in various lakes or reservoirs throughout the southeastern United States seldom exceeds 0.3 bass/hr (Van Horn and Birchfield 1981; Farman, Nielsen, and Norman 1982; Chapman and Fish 1983; and Dolman 1991). Because total angler effort at a given site is ultimately determined by where the individual angler decides to fish, the key issue may be how this decision was reached. Was it based on first-hand experience or second-hand information? In contrast, some anglers that fish regularly in Guntersville Reservoir probably consider the decline in fishing pressure as a benefit.

Frequently, the underlying issue in angler opposition to aquatic plant control or management is the amount of surface coverage of aquatic plants. Presumably, there is an optimum level of coverage, but this level probably varies widely depending upon the type of water body. Also, any systematic determination of optimum plant coverage in mainstream impoundments of large rivers, such as the Tennessee, is usually confounded by dynamic changes in water discharge and quality that can affect both plants and fish. Likewise, an optimum level of aquatic macrophytes for the fish community in Guntersville was not determined in the present evaluation, given the narrow range of plant coverage during 1990-1993 and the unexpected and rapid decline in plant coverage as grass carp were being stocked. Based on the current stable condition of the largemouth bass population, recent observations by Canfield and Hoyer (1992) in Florida may be applicable to Guntersville Reservoir. They suggested that when considering the total fish population (community), a moderate macrophyte coverage of 15 percent seems to preclude the probability of any adverse fisheries problems. However, from this survey of 60 lakes, they also found no relation between the standing crop of harvestable largemouth bass and the percent area covered by aquatic macrophytes.

Although the joint agency study (JAGP) officially concludes at the end of 1994, fisheries and aquatic macrophyte assessments will likely be continued at some level, given the major sport fishery in Guntersville Reservoir, other multiple-use conflicts, and the controversy surrounding the use of grass carp.

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Orange and Lochloosa Lakes Case Histories

History of Hydrilla Control in Orange and Lochloosa Lakes

by
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Introduction

Orange and Lochloosa lakes have long been noted as top fishing locations in this country, supporting a wide variety of common wildlife and endangered species. Both Orange and Lochloosa lakes are designated as Class III Outstanding Florida Waters by the Department of Environmental Protection and as Fish Management Lakes by the Florida Game and Fresh Water Fish Commission.

Overgrowth of hydrilla in Orange and Lochloosa lakes has been a threat to the economic and environmental health of the area since 1976.

Milon et al. (1986) indicated that regional economic activity attributed to Orange and Lochloosa lakes was over 10 million dollars. Colle et al. (1987) found that an 80-percent coverage of hydrilla on Orange Lake resulted in an 85-percent reduction in angler utilization and a 90-percent loss in revenue.

Milon's study also concluded that aquatic plant management was a significant contribution to the local economy of the community associated with these lakes. Aquatic plant management is an effective tool to improve fish and wildlife habit and also provides recreational and economic benefits to the region.

Orange Lake

Orange Lake is located southeast of Gainesville, FL, in Alachua County with a drainage area of 110 square miles. The lake has a shoreline perimeter of approximately 25 miles.

The lake receives water flow from Lochloosa Lake by way of Cross Creek and from Newnans Lake through the River Styx. The only surface drainage from Orange Lake is Orange Creek, a tributary of the Oklawaha River. Principal water losses are the result of evapotranspiration and downward drainage through the sediments and solution cavities in the lake.

The bottom of Orange Lake is predominantly unconsolidated organic matter over a clay layer covering limestone.

The water level in Orange Lake has a maximum range of 11.4 ft, ranging from a high of 61.2 FASL to a low of 49.8 FASL. This variation in stage results in lake coverage range of approximately 3,500 to 16,000 acres and volume range of 5,000 to 100,000 acre ft. The average stage for the last 30 years was 57.5 FASL, which would result in an acreage of 12,500 and 68,000 acre-ft of water.

Digitized aerial photographs indicate that emergent vegetation in Orange Lake has increased from a 34-percent coverage in 1964 to a 58-percent coverage in 1991. Major vegetation changes occurred in 1976-77 when spatterdock increased in the West Arm, in 1978 when a 4-ft increase in water levels redistributed floating islands on Orange Lake, and in 1991 when extremely low waters provided for a 700-acre increase in cattails.

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Algae blooms and influx of dark tannin-stained water have resulted in Secchi variations in the lake from 2 to 8 ft.

Lochloosa Lake

Lochloosa Lake is located southeast of Gainesville, FL, in Alachua County with a drainage area of 1,100 square miles and a shoreline perimeter of approximately 12.5 miles.

The lake receives water inflow from Lochloosa Creek and several minor creeks that feed into the marsh on the north end of the lake. The largest portion of the surface outflow is through Cross Creek into Orange Lake, with a small amount of outflow through Lochloosa Slough on the southeast side of the lake. During large storm events, water will occasionally switch directions and flow from Orange Lake to Lochloosa Lake through Cross Creek.

The bottom of Lochloosa Lake is predominantly unconsolidated organic matter over a clay layer covering limestone.

With the exception of extreme lows, Orange and Lochloosa lakes are usually at or near the same stage. Lake levels can vary approximately 7.5 ft from a low of approximately 54.0 FASL to a high of 61.5 FASL. This has resulted in a acreage variation of 5,700 to 10,500 and volume variation of 30,000 to 80,000 acre ft. The average stage for Lochloosa for the last 30 years is also approximately at 57.5 FASL, which would result in an average of 8,300 surface acres and 55,000 acre ft of water.

The shoreline gradient of Lochloosa Lake (1:65) is approximately three times steeper than Orange Lake (1:170), which is a primary factor limiting the growth of emergent plants in Lochloosa Lake, resulting in cypress surrounding the lake instead of large areas of emergent plants. Digitized aerial photographs indicated emergent vegetation occupied only 3.5 percent of the main lake in 1964, increasing to a 6.5-percent coverage by 1991. During this period, there was a slight decline in spatterdock coverage, with the increase in coverage the result of other emergent species such as cattails, pickerelweed, and primrose.

Algae blooms and influx of dark tannin-stained water have resulted in Secchi variations in the lake from 0.6 to 7.0 ft.

Orange Lake Hydrilla Management

Hydrilla was first reported in January of 1974 in the Cross Creek vicinity. It was probably introduced at the Marjorie Rawlings boat ramp 1 or 2 years previously. Insufficient funding limited hydrilla control operations to 10 acres of treatment at the mouth of Cross Creek and Marjorie Rawlings boat ramp. By the end of 1974, hydrilla increased to approximately 50 acres.

Hydrilla levels increased to 270 acres in 1975, but few people were concerned about its presence or the five-fold increase in coverage. Control measures were limited to 50 acres of boat trails and 8 acres of test plots, using endothall products and Cardi.

The hydrilla population exploded in 1976, covering 90 percent of open water, including most of the spatterdock areas, with a total coverage of 8,000 acres by the end of the summer. The first funds specifically for hydrilla control were provided by Alachua County, Marion County, and the Department of Natural Resources. A total of 150 acres of hydrilla were treated in the form of boat trails. An additional 49 test plots were tested for hydrilla control using Cardi, Karmex, Diuron, 2,4-D, Diquat, Hydrothol 191, Aquathol, Asulox, Komeen, Cutrine, and System L.

A harvesting program was conducted in the fall of 1976 through the fall of 1977 by the U.S. Army Corps of Engineers. A total of 160 acres of hydrilla were harvested in the form of boat trails and small fishing areas.

An additional 95 acres of herbicide control was conducted to assist in keeping boat trails open. Seven additional test plots were

conducted on hydrilla to determine if there was any difference in efficacy between existing approved products under the particular water quality conditions of Orange Lake. The conventional herbicides containing diquat, endothall, and copper resulted in a minimum of 95-percent control. Since Hydrothol 191 provided equivalent control to other products and was more economical, it was selected for treatment of boat trails.

In 1978, a tremendous decline in hydrilla occurred in Orange Lake as the result of a sharp rise in water level of over 4 ft and persistent, blue-green algae blooms. Hydrilla coverage was reduced to approximately 500 acres, allowing the hydrilla control program to be suspended for the year.

Low levels of hydrilla in 1979 limited the demand for control measures to providing access around the shoreline at fish camps and boat ramps.

Although funding increased substantially in 1981, hydrilla acreage increased to 6,000 acres in 1982. At that time, neither the funding nor the technology was available to control the large acreage of hydrilla with herbicides or mechanical harvesting.

In January of 1982, the first testing of the experimental herbicide fluridone was conducted in Orange Lake. Several small applications over the next 6 months, combined with a gradual 3-ft increase of water level from January to June, reduced hydrilla levels to 1,150 acres.

Even with limited funding, the hydrilla coverage was steadily reduced during the period of 1982-90, with coverage exceeding 30 percent for only a 2-month period in 1988.

A 45-percent increase in treatment area in 1990 reduced hydrilla coverage to 6 percent of the lake in June of 1991. With no funding for fluridone treatments in 1991, hydrilla levels increased to 14-percent coverage of the lake by October.

In 1992, hydrilla coverage increased to 4,100 acres or 67 percent of open water. A 500-acre fluridone application reduced hydrilla coverage by 800 acres.

Hydrilla coverage increased in 1993 to a maximum of 5,100 acres in August, but declined by 1,500 acres by January of 1994 as a result of a 920-acre fluridone application.

An 850-acre fluridone application was conducted in March of 1994 to further reduce hydrilla populations in the lake.

Improvements in the hydrilla management program are obvious when conditions and hydrilla populations in 1991 are compared with those in 1976. Since the lake was 2 ft lower in 1991, it would be expected that hydrilla's potential would equal or surpass 1976 levels. However, 1991 hydrilla coverage was only 6 percent of the lake as compared with 63 percent in 1976.

Lochloosa Lake Hydrilla Management

Hydrilla was first detected in Little Lochloosa in May of 1975 originating from fragments scattered up Cross Creek by boat traffic.

From 1975-1977, the hydrilla management consisted of treating all known areas of hydrilla with contact herbicides such as granular endothall and diquat invert application.

In 1978, hydrilla coverage had increased by 700 percent over the previous year, making it necessary to redirect the management program to provide navigation trails, boat ramp access, fish camp access, and homeowner access along the eastern shoreline. Management programs for 1979 and 1980 were directed at providing access as in the previous year.

In 1982, 7 years after hydrilla's introduction, the population covered 65 percent of open water of Lochloosa Lake. In comparison, 90 percent of open water of Orange Lake was

covered by hydrilla in only 2 years. Early management programs and the steeper gradient of the shoreline were principal factors in the slower rate of hydrilla domination in Lochloosa Lake.

The first use of fluridone for hydrilla control in Lochloosa Lake was conducted in 1982. Unknown at the time, the application was conducted under ideal lake conditions, a 3-ft increase in water levels from January to June. This program resulted in the control of 2,000 acres of hydrilla. Additional control measures in 1983 reduced hydrilla coverage by 1,300 acres. As in 1982, water levels were high at time of application (58 FASL) and rose to over 59 FASL from April to June.

In 1984 and 1985, water levels gradually decreased to a low of 55.8 FASL in June of 1985, allowing hydrilla to dominate 79 percent of the lake. Fluridone was available to treat only 25 acres for 1984 and 1985 under the experimental use permit. With declining water levels and no technique available to control large areas of hydrilla, the problem became serious.

In the spring of 1986, fluridone received full registration. Fortunately, water levels in 1986 had increased 2 ft over the previous June, increasing the potential for hydrilla control with the May fluridone program. From May to November, the hydrilla population was reduced by 3,700 acres or 82 percent.

Water levels rose again in the spring of 1987 to a high of 60 FASL in April, which again improved the effectiveness of the fluridone management program. The treatment resulted in the control of over 2,700 acres of hydrilla, of which the majority was composed of newly germinated plants.

Since hydrilla control funding was limited and there were less than 700 acres of hydrilla present in the lake, management was reduced to the control of 60 acres with fluridone and 110 acres with contact herbicide in 1988. In conjunction with declining water levels, the hydrilla population expanded to cover 85 percent of Lochloosa Lake from May to October. Management with fluridone at that time of year would have resulted in severe and unnecessary destruction of beneficial native vegetation in the lake.

Since the 300-acre fluridone treatment of 1986 was so successful in controlling hydrilla, the involved agency representatives and Center for Aquatic Plants surmised that good control of hydrilla would be obtained with a 250-acre application in 1989, while reducing expenditures. Unfortunately, it could not be predicted that water levels were going to drop 1.7 ft from March to August nor was it known at this time that treatment acreage required for large-scale management increases with dropping water levels and topped out vegetation. The unexpected result was the control of only 410 acres of hydrilla.

With water levels in 1990 continuing to drop, hydrilla topped out and covered 88 percent of the lake. The management program was increased by over 200 percent, which resulted in control of 2,000 acres of hydrilla.

In 1991, management operations were reduced 20 percent because of the previous year's results, and most of the hydrilla was not to the surface as in the previous year, making it more susceptible to the fluridone. A total of 2,600 acres of hydrilla were controlled by October.

In 1992, a total of 400 acres of hydrilla were treated with Sonar. This program was the least effective fluridone management program in Lochloosa. This was probably due to the combined factors of low water at time of treatment, dropping water levels after application, and a strip application that was spread out over a 55-day period. Only approximately 100 acres was controlled, with a double in total hydrilla coverage during the remainder of 1992.

The 1,200 acre 1993 application was the largest fluridone application ever applied to the lake. The program was conducted over a 2-week period, and the applications were not

spread out or strip treated as in 1992. The treatment provided excellent control, reducing hydrilla to a 12-percent coverage of the lake. With fluridone concentrations exceeding 5 PPB for over 5 months, nontarget damage was observed on native species, mainly spatterdock.

In the spring of 1994, hydrilla had regrown to cover 40 percent of the lake, with the majority of the plants less than 1 ft tall at time of herbicide application. A 900-acre fluridone application reduced hydrilla to 3.5-percent coverage by May.

From 1989 to 1994, total acreage of emergent plants have remained constant at 7.5 percent of the lake. Although this figure has not changed, there was a change in species composition. During this period, approximately 60 acres of bonnets were killed indirectly by the hydrilla control program; at the same time, there was a 65-acre increase in pickerelweed, cattail, knotgrass, and primrose willow in other areas of the lake.

The following graphs provide annual acreage estimates for hydrilla remaining after each growing season and the total number of acres of control during each year for Orange and Lochloosa lakes, respectively.

Summary

Management with mechanical harvesting

Harvesting with the Aqua-Trio system was expensive, costing $460 per acre ($1,078 in 1992 dollars) in Orange Lake. It was also slow, taking approximately 8 hr to harvest 5 acres of hydrilla. Frequency of harvesting required to keep boat trails open for the year varied from one to three cuttings per year, depending on the specific trail location. To achieve the same control as the 1993 Lochloosa fluridone treatment, a mechanical harvesting program would have required the use of six harvesters for a period of 4.5 months at a cost of approximately $4.47 million.

Harvesting in Orange Lake (Haller et al. 1980) resulted in a 32-percent loss of fish by numbers and 18-percent loss of fish biomass. This would translate to a monetary replacement value of fish harvested at over $2,400 per acre.

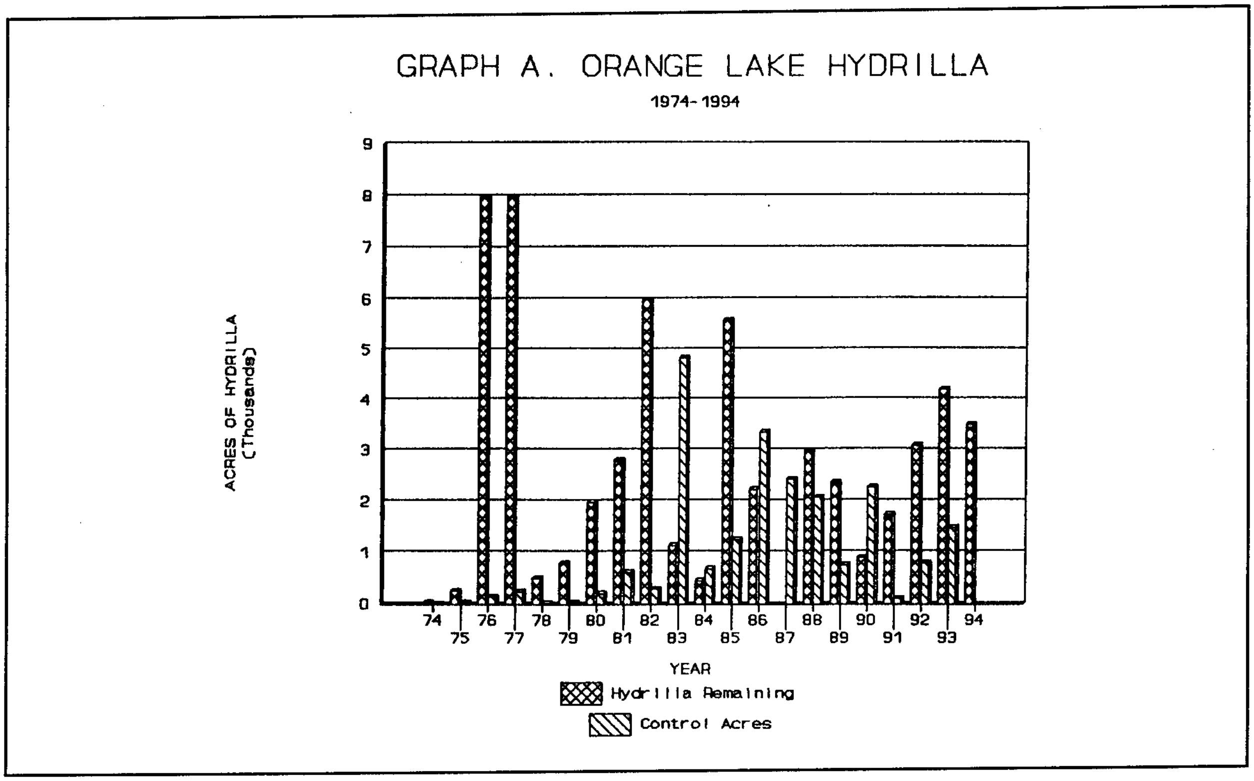
Management with contact herbicides

Since 1974, the contact herbicides diquat dibromide, endothall, and copper complexes were used in Orange and Lochloosa lakes for hydrilla control. These products were used during the first few years to slow down the spread of hydrilla in these lakes and have continued to be used to the present to provide boat trails, access area around ramps, and small fishing areas of less than 5 acres. Most contact applications resulted in 1 acre of control per acre of treatment, with occasional treatments providing an additional 25-percent control over the treated acreage. The frequency of control to keep any one area open varied from one to three applications per year. A total of 3,187 acres of hydrilla in Orange Lake and 1,170 acres in Lochloosa Lake were controlled with contact herbicides in the last 30 years.

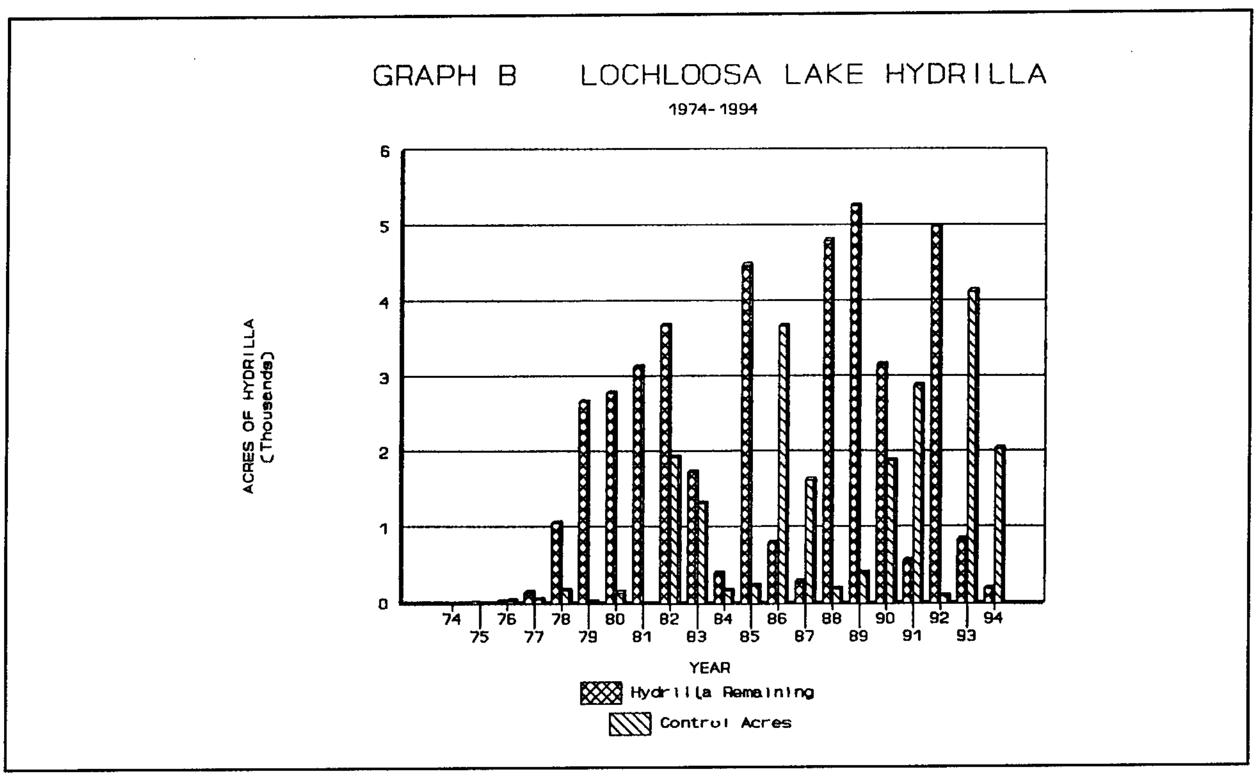
Management with fluridone

Since 1982, fluridone (Sonar) was utilized for control of large areas of hydrilla infestation. Fluridone was the first aquatic herbicide available that could provide large areas of hydrilla control and, in many cases, at a considerable savings over previously used contact herbicides and mechanical harvesting. With proper timing of herbicide application, the product could remove large areas of hydrilla from a lake with only minor damage to native aquatic plant species. Lower toxicity to fish and other nontarget organisms were additional benefits from using fluridone as a management tool.

Although most fluridone applications provided excellent control of hydrilla, the actual



Orange Lake Hydrilla



Lochloosa Lake Hydrilla

areas of control were many times located in areas other than the application sites.

As seen in the following table, the amount of control achieved per acre of treated hydrilla in Orange and Lochloosa lakes varied considerably from one year to the next.

Hydrilla Control with Fluridone, 1982-93		
	Orange	Lochloosa
Total Acres of Control	20,454	20,547
Total Acres Treated	3,572	3,606
Ave. (Acres Control/ Treated Acre)	5.7	5.7
Range (Acres Control/ Treated Acre)	1-24	0.25-30
Average Cost Per Acre	$114	$114
Range in Cost Per Acre	$27-$650	$22-2,600

Since the amount of hydrilla control was not directly related to the amount of herbicide applied, it was apparent that physical and/or biological factors were affecting the degree of hydrilla control.

Preliminary results using multilinear regression indicate that the following factors can contribute to increased efficacy of fluridone applications for hydrilla control on these lakes:

a. Calm weather conditions during and after herbicide application.

b. Herbicide treatments conducted over a short period of time, no split applications.

c. Normal- to high-water levels (>57 FASL) at time of herbicide application.

d. Increase in water level after herbicide application.

e. Hydrilla plants actively growing and not to the surface.

f. Low water clarity.

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Probable Grass Carp Stocking Scenarios

by
William T. Haller[1]

Introduction

Aquatic plant managers are now in a dilemma. Historically, aquatic weed control whether by mechanical, chemical, or physical means has provided only temporary relief from aquatic weed infestations. Classical insect biocontrol is by its nature species specific and has not altered the overall structure of our aquatic ecosystems. In addition, the high cost of chemical and mechanical control has limited the extent to which these control methods are applied.

Consequently, fisheries and wildlife biologists have never been overly fearful of the aquatic plant managers ability to alter aquatic plant habitat on a long-term basis.

Grass Carp

Aquatic weed growth in Florida is more widespread than any other state because of the warm temperate to tropical climate and an abundance of shallow, naturally eutrophic water bodies. Authorities (Federal and State) have been managing water hyacinth since the 1890s and more recently have been faced with widespread expansion of the submersed weed hydrilla (introduced into the State in the 1950s). Hydrilla has become the major problem in Florida because of the high cost of mechanical or chemical control. In the mid-1960s, State agencies introduced and evaluated a *Tilapia* species for possible hydrilla control and by 1969 began to investigate the use of grass carp for control of submersed weeds.

In 1976, a grass carp symposium was held at the University of Florida, and the hope and fears of the participants regarding the use of this fish for weed control were evident.

It was hoped that the grass carp would prefer to feed on hydrilla, dislike and not significantly affect native submersed plants, have a life span of 5 to 10 years, minimally affect water quality, have no adverse effect on sportfish, and be easily removed from aquatic ecosystems.

On the other hand, many feared that the grass carp would reproduce, migrate or move great distances, remove all submersed vegetation, live for 20 to 30 years, create algae blooms and poor water quality, destroy sportfish production, and be impossible to remove from our waterways.

Over the ensuing 18 years, many of these hopes and fears have been addressed and answers attained. All were essentially covered in this symposium and proceedings. In the past two decades, well over 2,000 bodies of water (from 0.1 to 27,000 acres) have been stocked with grass carp in Florida, and research is continuing.

A review of plant populations and surveys of previously stocked lakes (>50 acres) in Florida indicates that in the majority of cases, the stocking of grass carp either totally eliminated submersed species or had no impact on them. Most biologists acknowledge that weed control will occur provided enough grass carp are stocked (and survive) to overcome the growth of the vegetation. The problem arises when the fish are stocked; there is no way known to determine survival 1 hr, 1 month, or 1 year after stocking. The answer to this problem some biologists believe is to stock with low rates of fish (3 to 5/acre?) and gradually increase stocking rates until weed control is achieved.

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Two Possible Scenarios

Given a 5,000-acre landlocked prime sportfishing lake plagued with hydrilla for 10 years. The annual expenditures for hydrilla control with herbicides average $300,000/year. It is decided to stock the lake next year following herbicide treatment with four grass carp per acre at a cost of $5/grass carp (5,000 acres × four fish × $5/fish = $100,000).

Scenario 1

The herbicide is applied in February and March, and excellent results are attained. The 20,000 grass carp are added in May, and heavy rains bring water levels for the summer up an additional 4 ft. Grass carp survival is near 100 percent, and the high water prevents hydrilla growth in the center 60 percent of the lake. The next year, water levels return to normal, and there are twenty thousand 10-lb grass carp in the lake capable of eating 10 to 15 lb of vegetation per day. Hydrilla tubers germinate and sprout above the hydrosoil and are immediately grazed by roving grass carp. By late summer of this second year post-stocking, hydrilla cannot regrow; it disappears as the grass carp feed on the 50 acres of maidencane in the lake. By the third year, the regrowth potential of hydrilla is further reduced because of less hydrilla tubers surviving (none produced now in 2 years) and the grass carp consume all submersed species and greatly reduce emergent species. After 5 years of nonexistent vegetation in the lake, sportfishing declines and brush piles are added to the lake to bring fish populations back into balance (if possible in a 5,000-acre lake). The lake remains plant free for 20 years, and revegetation is attempted to improve sportfishing, but is unsuccessful because there is a residual of 1/2 to 1 grass carp per acre. Finally, after 25 to 30 years, revegetation is successful, but hydrilla is reintroduced and more grass carp are added and Scenario 1 repeats itself.

Scenario 2

As in Scenario 1, the herbicide is applied in the spring of year 1, grass carp are added at four per acre as before, but because of little hydrilla (cover) in the lake, predation of the fish is heavy (60 percent) and only 1 to 1.5 grass carp/acre survive until year 2. Hydrilla regrowth is rampant by the second summer as low water promotes its growth and the grass carp have no impact on the vegetation. Herbicide treatment is again conducted in the spring of year 3, and an additional five fish to the acre are stocked. Predation is once again high. The herbicide did not work well, and the following summer (year 3), the lake is again covered with hydrilla. Herbicide is applied in the spring of year 4, and ten grass carp/acre are stocked. Water levels rise, the herbicide treatment is very effective, and now we return to Scenario 1, but with 8 to 12 surviving grass carp per acre, not the 4 as stocked in Scenario 1.

Discussion

Under both scenarios, we end up with a lake with very little emergent vegetation, possibly some *Nuphar* survives, and no submersed vegetation for 15-20-25 years. The annual cost of herbicide treatment is saved. If the lake historically supported a $3 million/year sportfishery, will it continue to after 5-10-15 years with no or little vegetation? Our hopes are that the sportfishing will continue and water quality will not be adversely affected. A pertinent question was covered in Lawson Snyder's talk on grass carp removal from Clear Lake. Why did the residents of Clear Lake want the grass carp removed after 15 years? Will the residents of this hypothetical lake want the grass carp removed after 15 years? Will we be more efficient at removal in 15 years? What is the importance of aquatic vegetation to sportfish production? Wildlife? As with stocking rates of grass carp (no magic number exists for all water

bodies), the management objections of large fish and wildlife management lakes are going to have to be well defined. There is no doubt that the grass carp can provide cost-effective weed control, and it is a strong and easy case to make. The question remains, however, that aquatic plant managers can now can provide long-term weed control, but do we know with a reasonable degree of confidence the long-term effects of an ecosystem with no submersed vegetation for 10-15-20 years?

Comments on the Symposium

by
David Brakhage[1]

After attending the 3-day symposium on grass carp, I would like to offer some comments about the use of grass carp to manage aquatic plants from my perspective as a waterfowl biologist.

There was limited participation by wildlife biologists and wetland ecologists in the symposium. This seems to be the case on the grass carp issue in general, which is unfortunate. A key factor for us all to keep in mind when addressing any aquatic plant management issue is that wildlife have a significant stake in the outcome of aquatic plant management decisions. Grass carp are a partially selective herbivore that have the capability to entirely eliminate aquatic vegetation from a system if present in sufficient numbers. Although they exhibit a preference for hydrilla, they will readily consume several other species of native plants that have been demonstrated as desirable for fish and wildlife. Concern over the use of grass carp relates to the possibility of overcontrol of vegetation and the elimination of not only the target plants, but desirable plants as well. The debate continues over the appropriate amount of vegetation needed in a lake system, if any, to maintain healthy fisheries. There is no debate, however, where wildlife are concerned. Desirable aquatic macrophytes are absolutely essential to wildlife species that rely on wetlands. These species include more than just ducks, too. Numerous other birds, reptiles, amphibians, and mammals rely on wetlands for their survival. The majority of these require aquatic macrophytes, and the boundaries of their desirable habitat is generally delimited by the occurrence of aquatic macrophytes within a system.

Stocking sufficient grass carp to quickly eliminate an established stand of hydrilla will invaribly lead to overcontrol of native plants once that hydrilla has been eliminated. The only way to avoid this is to either use grass carp initially when hydrilla remains at low levels or first knock back the standing crop of hydrilla with herbicides before grass carp are stocked. This latter approach may not be successful with hydrilla because massive growth potential may still exist in the system because of the roots and tubers that remain even though the standing biomass has been reduced. We have had limited success in achieving and maintaining the stocking rate necessary to control nuisance plants without impacting desirable plants. Our ability to learn about the appropriate stocking rates to reduce or eliminate hydrilla and achieve maintenance control is hampered by our inability to know what the actual stocking rate is at any given time. Some reliable method to remove grass carp must be perfected to allow their use in a safe manner and to ensure that overcontrol does not occur.

The desire to use grass carp in the face of uncertain effects is largely a matter of economics. Grass carp are less expensive than herbicides when the goal is total plant eradication. Grass carp may also be less expensive than herbicides when the goal is to maintain some level of aquatic plants in the system. An integrated management system combining grass carp and herbicides has appeal, but it is not as predictable or as safe as using herbicides alone. Furthermore, an integrated approach may not necessarily be less expensive than chemicals alone. Although required less frequently, herbicide treatments are still needed under an integrated system, and additional grass carp frequently need to be stocked to maintain the appropriate level of control. There may also be the additional cost of grass carp removal if the stocking rates get too

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high. The use of contact herbicides rather than Sonar to fine-tune the stocking rate of grass carp per vegetated acre seems to make sense because contacts give a more predictable result than the systemic herbicide Sonar.

Clearly, based on the papers presented during this symposium, there is no cookbook answer available for the use of grass carp to manage aquatic plants. At least right now, it appears that every system will have to be treated separately with a commitment to monitor the effects of the control program and modify the approach as needed to maintain appropriate levels of vegetation in the system. Quality research should continue to identify safe and effective ways to use grass carp for aquatic plant management. Grass carp have the potential to be used effectively in an overall program of aquatic plant management. I'm very reticent about using this fish widely before a technique is perfected to remove the fish when needed. I also caution aquatic plant managers not to succumb to the short-term interests of individual property owners or lake-user groups, but maintain an appropriate perspective on what is best for lake ecosystem and all its dependent organisms over the long term. Diploid grass carp should never be used under any circumstances, and every reasonable effort should be made to eradicate them from our nation's waters. I encourage the U.S. Fish and Wildlife Service to take action to completely ban the use of reproductively capable grass carp for aquatic plant management. Responsible and consistent use of grass carp requires a clearly stated policy and a coordinated planning effort where all stakeholders are appropriately represented.

Summary Comments

by
J. L. Decell[1]

Currently, there are approximately 30 States that permit the use of the grass carp. A lot has changed since the 1970s when only a few States were actively involved in grass carp stocking. The "bottom line" concern in those days was reproduction. Fisheries people wanted a nonreproducing fish—everyone wanted guarantees—both ways.

They finally got their wish when the triploid was developed, and guess what? The "bottom line" changed! It shifted from wanting a guarantee of nonreproduction to a guarantee that every fish in the population is a triploid. Well, they've almost gotten that wish! Finally, the concern has shifted to a broader ecological element—that of habitat destruction. While everyone might argue that this was really the concern behind their respective demands, I would argue that it was certainly not at the forefront, where it should have been all along. Are we finally arriving? Have we found the enemy?

Someone used the phrase, "we cannot guarantee." Someone else made reference to "a silver bullet." The only silver bullet I know of is marketed by the Joseph Coors Brewing Company—and I'm not sure exactly what it guarantees! Sometimes, without realizing it, people state that they want a guarantee when they have reached the point of being inadequately informed to pursue the argument at hand.

Based on several of the presentations, it is obvious that we have now begun to study removal techniques. For emergencies and extenuating circumstances, this is good. The basic question, however, is why? Why do we want to remove them? Without a rationale that stands on its own merit, we incur the risk of admitting to overstocking. We should continue to explore the development of methods for removing the fish. It should not be done to develop a "secondary management" step or a "safety net" to make up for a lack of adequate consideration during the prestock planning.

We should constantly remind ourselves, and those that we advise, that we are using the fish to manage the plants—we are not managing the fish any more than we manage a herbicide after it's placed in the water. Using the grass carp for plant management requires that you anticipate the result you want to achieve, and based on a knowledge of the fish and the system interactions, determine what "amount" of fish will provide that result. You then apply the correct "amount," at the optimum times and places, monitor the initial system response, and if necessary, at some appropriate later time, supplement the application. This supplement is additive—not a reduction. If this process sounds familiar, it's because it's what happens regardless of the control method. With the exception of a mechanical harvester, you do not remove a control agent from an aquatic system!

We should not stock grass carp because we can remove them—and we should not avoid stocking them because we cannot remove them. Both of these cases are the wrong reasons for stocking or not stocking.

In my opinion, this is a diversion to avoid having to rationally deal with the complexities of managing a natural system.

From an overall perspective, as environmentally sensitive aquatic plant managers operating in our respective roles as stewards, we should not be a proponent of any one control method. We should only be a proponent of

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the proper use of the appropriate method, given a clear, and as complete as possible, assessment of the conditions in which we must operate.

In that context, I would submit that nearly everything you have heard here during these 2 days could have been said about the use of any other type of control.

I would like to share a perspective that we should all embrace now and more importantly, for the future. The reason we are in business should be obvious! We are trying to restore man's valued uses of the water bodies by controlling a plant population that has interfered with those uses. It is not the population of the plant that is the problem—it's simply that it is occurring at some level, in some place, that we want to use for our particular purposes. There are, however, trade-offs that must be made, and therein lies the perspective.

If we are to manage and not just control; if we are to be judged to do our business in an environmentally compatible manner, then we must recognize that we are, in fact, managing habitat.

The use of the grass carp is not a question of considering available technology. Nor is it a consideration of viability. It is simply a consideration of acceptability.

When the challenge is presented, if we spend our time trying to make a case based on lack of technology or viability, we will not be making an investment to find a solution to the problem at hand. Coming to this recognition won't suddenly provide us with additional control methods, but it won't take any from us either.

It can, however, commensurately cause us to rethink and restructure our objectives. This will begin to influence the way in which we apply the knowledge at our disposal and will subsequently change the way in which we select and apply the methods we use. That will have to take place in order for us to manage and not simply control.